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Supporting ecosystem services with conservation agricultural approaches

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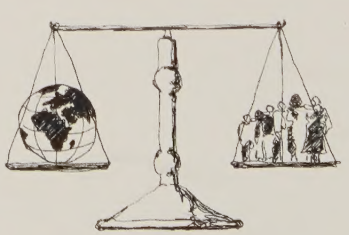

"Sustaining Earth and its People"

Scales image: Design, George Eisele, Lincoln, NE; artwork, Pat Scullion, ASA/CSSA/SSSA, Madison, WI, USA

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Editorial: Introduction to themed section—supporting ecosystem services with conservation agricultural approaches

Ecosystem services are the properties and processes of the natural world that contribute to the well-being of plants, animals and humans in a holistic and global context. For too long, members of the agricultural community have been solely focused on the provision of food, feed and fiber. Of course, this essential human innovation has provided hugely important products engineered for mass production that serves all of human society. However, agriculture and the ecosystems in which it is practiced provide numerous other services that are becoming increasingly relevant to its very survival and the survival of humans that it supports. Several recent reviews have expressed the need to balance production and environmental goals to create a sustainable future[1–3].

Ecosystem services have been categorized into four key components, including supporting, regulating, provisioning and cultural services (Fig. 1). Agricultural landscapes have a large and sometimes dominating influence on several essential ecosystem services, including biomass production (i.e., food, feed, fiber and fuel production), air purification and climate regulation through greenhouse gas exchange, soil formation and retention, water cycling, nutrient cycling, wildlife habitat provision and aesthetic experience.

Biomass production is a key goal for agricultural systems in the form of high-quality food products, fodder production for consumption by ruminant animals and wood and fiber as raw materials for shelter, manufacturing and textiles. Renewed interest has emerged in deploying agricultural systems for the production of biofuels to meet growing energy demands and reduce reliance on fossil fuels[4].

Air purification and climate regulation through greenhouse gas exchange is a key characteristic of all vegetated landscapes, including agriculture. The ability of plants to fix CO_2 and emit O_2 is a vital ecosystem process. Agricultural lands contribute to drawing down CO_2 in the atmosphere via photosynthesis, but also storing some of the fixed carbon in soil as organic matter. Historically, however, with exploitive inversion tillage practices, soil organic matter has been severely lost in many agricultural soils.

Soil formation and retention is an ecosystem service that is too often neglected by historical agriculture, resulting in severe erosion and loss of fertility of agricultural landscapes. However, regenerative agricultural practices can maintain and build rich, new soil that can be utilized for generations to come.

Water cycling, water quality and water infiltration are processes and properties greatly influenced by agriculture. Limited production focus on short-season crops without sufficient cover cropping and diverse crop rotations has often led to inefficient water utilization, loss of nutrients during non-growing periods, and poor surface characteristics through intensive tillage that leads to surface crusting and lack of stable aggregation that limits water infiltration.

Nutrient cycling is vitally important in agricultural systems, since nutrients are often the key limiting factor that separates naturally occurring ecosystems with closed nutrient cycling from agricultural systems that have enormous harvest outputs, which must be replenished with nutrient inputs either through cultural manipulations of cropping systems in time and space or through technological interventions of organic and inorganic nutrient applications.

Wildlife habitat provision in agricultural ecosystems is too often viewed as either an ancillary benefit if it occurs, not important in the grand scheme of biomass production goals, or simply not a priority in designing a sustainable agricultural landscape. Wildlife habitats can be enhanced with permanent surface cover, a landscape mosaic of managed crops and unmanaged natural and riparian areas, limited soil perturbations and limited pesticide applications.

Aesthetic experience is a cultural feature that is increasingly appreciated by society, given our desires and propensity to see the beauty of the world and its people. Should not our local and regional landscapes be something to be appreciated and highlighted for their intrinsic natural beauty?

Ecosystem services can be either degraded or enhanced with contemporary agricultural practices. Degradation of water sources can occur with aggressive soil management practices that cause soil loss and pollution of waterways with excessive sediment and nutrients. Waterways in agricultural settings can be further polluted with 'chemicals of emerging concern' when excessive therapeutic treatments are used, such as antibiotics and hormones from animal waste pathways. Degradation of air can occur with dust and chemicals from exposed agricultural fields without sufficient cover to maintain integrity against wind erosion. In addition, global air quality can be impaired with rising greenhouse gases emitted through carbon oxidized from soil as carbon dioxide, through nitrous oxide emitted from soil with excessive nitrogen

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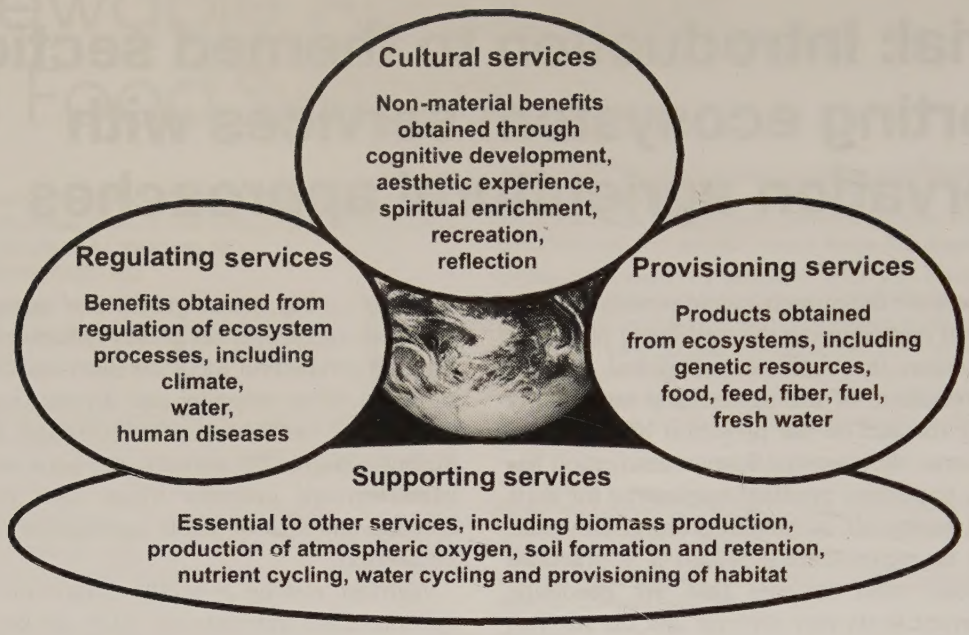
Cultural services
Non-material benefits obtained through cognitive development, aesthetic experience, spiritual enrichment, recreation, reflection
Regulating services
Benefits obtained from regulation of ecosystem processes, including climate, water, human diseases
Provisioning services
Products obtained from ecosystems, including genetic resources, food, feed, fiber, fuel, fresh water
Supporting services
Essential to other services, including biomass production, production of atmospheric oxygen, soil formation and retention, nutrient cycling, water cycling and provisioning of habitat

Figure 1. Categories of ecosystem services provided by nature, of which agricultural systems are a vital human component. Adapted from Millennium Ecosystem Assessment[5].

application to agricultural fields, and through methane emitted by ruminant animals and cultivation of wetland soils. Degradation of soil can occur with excessive tillage and non-renewing agricultural practices that deplete soil organic matter and cause poor condition in the chemical, physical and biological properties of soil.

Innovative farmers and agricultural researchers continue to explore a diversity of management approaches to renew natural resources within their local and regional areas. One categorization of agricultural practices designed to protect and invigorate soil, water and air quality and increase productivity and rural livelihood is conservation agriculture. Conservation agricultural systems are diverse, but have three key principles that can be used and modified to improve ecosystem services: (1) minimizing soil disturbance, (2) maximizing soil surface cover and (3) stimulating biological activity. Conservation agricultural systems were first developed with the development of conservation- or no-tillage management of croplands. However, it became increasingly evident that conservation tillage alone was not enough to elevate agricultural systems to a sustainable level. Crop diversity and crop rotations are needed to manage surface residues and weeds, cycle nutrients efficiently and utilize available water efficiently. Utilizing all available nutrient sources through integrated nutrient management or organic and inorganic sources is viewed as necessary to achieving high productivity and maintaining environmental quality. Managing pests with an integrated approach is essential through timing and type of cultural practices, knowledge of pest life-cycle history, threshold management and targeted biocidal applications.

Conservation agricultural practices can have an enormous impact on improving water cycling, quality and infiltration, all of which have cascading effects on other ecosystem properties. Conservation agricultural systems aim to close nutrient cycling to mimic natural ecosystems by minimizing nutrient losses to the environment, but sustainably replenishing nutrient harvests. Improving soil biological activity to limit nutrient losses to the environment must be a priority. Another key goal is to close nutrient cycling more on the landscape level rather than only individually on the field or farm level. Better systems are needed to return nutrients transported off the farm via harvested products back to agricultural fields. Conservation agricultural systems focus on providing surface cover, not disturbing soil, and utilizing biological diversity to improve agricultural system performance, but which also helps improve ecosystem functioning and wildlife habitats. Spatial design of agricultural landscapes using riparian buffers, wildlife corridors, cover crops and a diversity of crops, animals and phenological diversity will require stakeholder cooperation at local, regional and state levels.

Climate change, food security, water quality, soil degradation and renewable energy are our contemporary grand challenges interlinked with regulating, provisioning, cultural and supporting services provided by ecosystems. Hence, greater appreciation is needed of the value of a diversity of conservation agricultural approaches to meet the growing interest of farmers, scientists and society in developing a more productive and healthier environment with robust ecosystem functioning, not only within traditionally cropped fields, but also in a broader view of

a spatially diverse landscape that includes grasslands, forests and wetlands.

A symposium was convened at the Annual Meetings of the American Society of Agronomy, Crop Science Society of America, and Soil Science Society of America in San Antonio, Texas in October 2011 to describe ecosystem services from conservation agricultural approaches. Noted experts expounded on various conservation approaches that can enhance ecosystem services derived from agricultural production systems. Presentations addressed specific components of agricultural systems, such as conservation tillage, crop diversity, pasture management, organic agricultural approaches and wetland management. Practical experiences on the farm, as well as insights from ecosystem modeling and opportunities through government programs, were explored to foster new ideas and new approaches toward making agricultural systems more robust and ecologically resilient within watersheds and regional landscapes. Four papers from this symposium are presented in this themed issue:

Lal describes benefits and limitations of no tillage to enhance ecosystem functions and services, including through soil carbon sequestration, biodiversity, elemental cycling and resilience of food production systems to natural and anthropogenic perturbations. The argument is made that undervaluing ecosystem services provided by conservation agricultural systems will jeopardize finite natural resources and aggravate disservices.

Liebman et al. use the results of three long-term field experiments in Iowa to illustrate that diversifying cropping systems can substantially reduce chemical and energy inputs. Additionally, strategic placement of perennial buffers on the landscape can greatly improve soil and water conservation, nutrient retention and biodiversity. Perennial forages can be harvested for production of biofuels and contribute to significant soil carbon sequestration.

Sanderson et al. describe how crop diversity can be used to improve ecosystem services from managed grasslands and integrated crop–livestock systems. Tradeoffs among ecosystem services are also discussed. Dynamic cropping systems can optimize crop and soil-use options to attain production, economic and resource conservation goals by using sound ecological management principles. Integrated crop–livestock systems may increase complexity of management, but also create synergies among the components to improve resilience and sustainability.

Cavigelli et al. outline how organic grain cropping systems can enhance ecosystem services compared with conventional grain systems. Specifically, soil carbon sequestration and nitrogen mineralization potential can be greater in organic systems with manure and cover crops than in no-tillage cropping. A need was recognized for more data to assess greenhouse gas emissions in organic systems. Expanding crop rotations to include greater crop phenological diversity, improving nutrient management and reducing tillage intensity and frequency were identified as ways to further enhance ecosystem services in organic cropping systems.

It is my hope that these papers will stimulate discussion and instigate new research and development into agricultural practices and systems to meet the urgent needs of society for food, feed, fiber and fuel on the one hand, and strong environmental integrity and quality on the other. Our needs today and in the immediate future must be balanced with the needs of our heirs far into the future—a necessary goal of renewable agriculture and food systems!

'Treat the earth well: it was not given to you by your parents, it was loaned to you by your children. We do not inherit the earth from our ancestors, we borrow it from our children.'—Ancient Native American Proverb.

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Enhancing ecosystem services with no-till

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Abstract

Ecosystem functions and services provided by soils depend on land use and management. The objective of this article is to review and synthesize relevant information on the impacts of no-till (NT) management of croplands on ecosystem functions and services. Sustainable management of soil through NT involves: (i) replacing what is removed, (ii) restoring what has been degraded, and (iii) minimizing on-site and off-site effects. Despite its merits, NT is adopted on merely ~9% of the 1.5 billion ha of global arable land area. Soil's ecosystem services depend on the natural capital (soil organic matter and clay contents, soil depth and water retention capacity) and its management. Soil management in various agro-ecosystems to enhance food production has some trade-offs/disservices (i.e., decline in biodiversity, accelerated erosion and non-point source pollution), which must be minimized by further developing agricultural complexity to mimic natural ecosystems. However, adoption of NT accentuates many ecosystem services: carbon sequestration, biodiversity, elemental cycling, and resilience to natural and anthropogenic perturbations, all of which can affect food security. Links exist among diverse ecosystem services, such that managing one can adversely impact others. For example, increasing agronomic production can reduce biodiversity and deplete soil organic carbon (SOC), harvesting crop residues for cellulosic ethanol can reduce SOC, etc. Undervaluing ecosystem services can jeopardize finite soil resources and aggravate disservices. Adoption of recommended management practices can be promoted through payments for ecosystem services by a market-based approach so that risks of disservices and negative costs can be reduced either through direct economic incentives or as performance payments.

Key words: natural capital, carbon sequestration, nutrient cycling, erosion control, residues management, biodiversity, food security, climate change

The importance of the moldboard plow, in conjunction with the use of tractors and mechanization of farm operations, in expanding the land area under cultivation and increasing farm productivity is of historical significance. However, the vulnerability of plowed soil to accelerated erosion, compaction, oxidation of soil organic matter, emission of greenhouse gases (especially carbon dioxide) and other environmental issues (i.e., loss of soil biodiversity) led to the development of no-till (NT) farming.

Keen[1] observed that the benefits of plowing were mostly attributed to weed control. The rationale of plowing was questioned by Faulkner[2,3] because of serious erosion during the Dust Bowl era, even when herbicides were not yet available. Despite the intense debate on plowing[4] or not[2,3], it was the discovery of herbicides after World War II that made NT farming a practical reality. Indeed, NT farming evolved in the late 1950s and early 1960s primarily to reduce risks of accelerated soil erosion[5,6] and non-point source pollution from row cropping in the US Corn Belt[7–14]. Experiments on direct drilling in Europe started in the early 1970s with the availability of paraquat[15]. The historical roots of moldboard plowing, which cuts and turns over the soil, lie in a gradual replacement of the ard (a scratch tool or a modified v-shaped stick) by a metallic tool during the 7th century[16,17]. The ard, developed for alluvial soils of arid and semi-arid climates of the Fertile Crescent and valleys/flood plains of the Indus and Nile, was not a suitable tool for the humid climates and fine-textured soils of northern Europe. Thus, the moldboard plow gradually replaced the ard. The moldboard plow used in the US was designed by Thomas Jefferson in 1784, patented by Charles Newfold in 1796, and marketed in the 1830s as a cast iron plow by a blacksmith named John Deere[16].

Any system that eliminates all pre-seeding seedbed preparation is called NT or direct seeding. Three principal components of NT farming are: (i) elimination of all pre-seeding tillage (i.e., primary, secondary and tertiary), (ii) retention of crop residues, and (iii) control of weeds by

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Table 1. Estimates of land area under conservation agriculture (adapted from Kassam et al.[25] and Friedrich et al.[26]).

	Land area (10^6 ha)	
Year	World	South America
1973–77	2.7	0.06
1978–82	2.7	0.2
1983–87	6.3	0.7
1988–92	11.4	1.4
1993–97	35.0	9.0
1998–02	66.8	19.9
2003–07	100.3	27.6
2008–09	106.5	27.9
2010–11	124.8	55.4

herbicides. Weed control is a major challenge, especially in soils of the tropics, in the presence of perennial weeds and for organic farming. Thus, use of cover crops and residue mulch is a strategic option. Establishing an aggressive cover crop can smother weeds, while improving soil structure, nutrient cycling and availability. Soil and water conservation, specifically erosion control, has been the principal advantage of stubble mulch[18,19] and NT farming[7,8,20–24]. The concept of NT farming was expanded during the 1990s to conservation agriculture, which is comprised of five components: (i) elimination of pre-seeding tillage, (ii) adoption of complex crop rotations, including cover crops and agroforestry, (iii) use of integrated nutrient management strategies, (iv) control of weeds and other pests by integrated pest management, and (v) conservation, harvesting and recycling of water.

Globally, NT farming has been adopted in about 125 million hectares (Mha) out of a total of 1500 Mha of cropland area (Table 1)[25,26]. The rate of adoption was very high during the first decade of the new millennium in commercial and large-scale farms of South America (Table 1). In turn, conservation agriculture in the 1990s evolved into sustainable land management during the 2000s. Sustainable land management is defined as a knowledge-based combination of technologies, policies and practices that integrate land, water, biodiversity and environmental concerns (including input and output externalities) to meet rising food and fiber demands, while sustaining ecosystem services and livelihoods[27,28]. Five components of sustainable land management are: (i) adoption of conservation agriculture principles, (ii) adoption of a judicious land use, (iii) adaptation to climate change, (iv) mitigation of climate change, and (v) enhancement of soil/ecosystem resilience.

Ecosystem Services

Ecosystem functions refer to the habitat, biological, or system properties or processes of ecosystems. Ecosystem goods (i.e., food) and services (i.e., water filtration and pollution abatement) refer to the benefits that humans derive from ecosystem functions[29,30]. Specifically, ecosystem services are the conditions and processes through which natural ecosystems sustain and fulfill human demands[31]. Principal ecosystem services, outlined in Fig. 1, include provisional (food, feed, water, etc.), moderatory (climate stabilization, water filtration, pollutant denaturing, etc.), ecological (elemental cycling, biodiversity, etc.), and aesthetical and cultural (recreational, spiritual, etc.)[32–34]. Numerous ecosystem services essential to human wellbeing are provided by the soil. However, there are important trade-offs and synergies that need to be objectively considered[35].

Soil is a four-dimensional (length, width, depth and time) geo-membrane at the lithosphere/atmosphere interface that is a medium for plant growth and moderator of all ecological processes essential to functioning of terrestrial and associated aquatic processes. Depending on the predominance of specific soil-forming factors[36,37], soils provide a range of ecosystem services across the landscape. Soil quality, defined as the capacity to perform ecosystem functions, depends on the activity and species diversity of soil biota, which transforms and recycles matter and energy. A gram of fertile soil may contain millions of microorganisms, including protozoa, algae, fungi and numerous other known and unknown species. Major ecosystem services of a soil are generated by numerous biogeochemical cycles driven by solar and chemical energy and operating at different scales, ranging from global (carbon, nitrogen and water cycles) to molecular (chemical transformations).

Land use and management strongly impact soil's ability to provide ecosystem services. Conversion of land from natural systems to agroecosystems, a necessity to support the human population and its ever-increasing demands driven by growing affluence, can provide, but also curtail, numerous ecosystem functions and services. Therefore, a sustainable strategy must be to minimize the adverse impacts of land-use conversion for production of feed, food, fiber, fuel, and other goods and services to meet human demands. It is precisely in this context that widespread, but prudent adoption of NT farming, conservation agriculture (CA), and sustainable land management can play a pivotal role.

Natural Capital of Soil and Ecosystem Services

Sustainable use of finite soil resources necessitates a thorough understanding of the role of soil processes in relation to ecosystem functions, including but not limited to human wellbeing. In this context, Robinson et al.[38] proposed the concept of 'natural capital' of soil within the framework of ecosystem services. A clear distinction in the use of terms is essential, because many terms have multiple definitions[39]. The term 'natural capital' is an

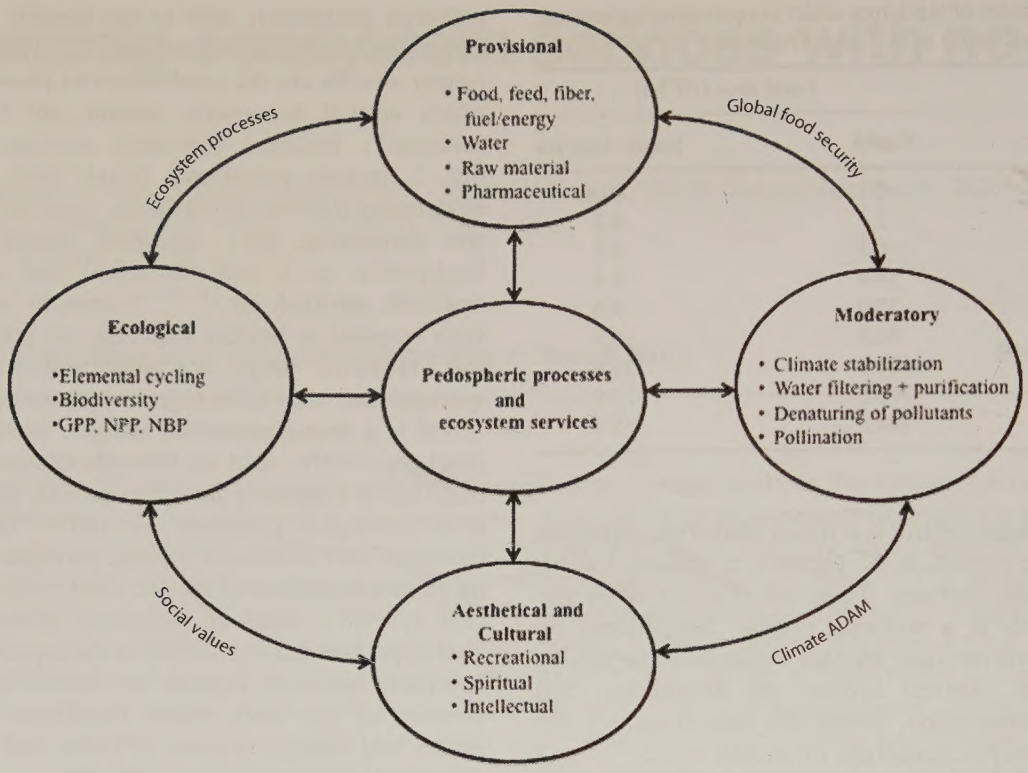

Figure 1. Types of ecosystem services.

extension of the economic idea of manufactured capital to include environmental goods and services. As an analogy, natural capital of soil would consist of its properties such as clay and soil organic matter contents, soil depth, water retention capacity, etc. Thus, ecosystem services from soil would include tangibles such as net primary production, agronomic yield, climate moderation through carbon sequestration, methane oxidation, etc. Accordingly, Robinson et al.[38] proposed that principal components of soil's natural capital would include: (i) mass (solid, liquid, and gaseous), (ii) energy (thermal), and (iii) organization or entropy. All three components have quantitative and qualitative attributes. Furthermore, sustainability of agroecosystems depends on strategic management that integrates the natural capital of soil across different scales from molecular to global. These are among several major ecological challenges that must be scientifically addressed[40]. Furthermore, apparent conflict among ecosystem services (i.e., food versus fuel and agronomic productivity versus biodiversity) must also be resolved. For example, mandating the use of biofuel in one place may increase emission of greenhouse gases at another due to land-use changes[41]. Implementation of 'The Billion-Ton Biofuels Vision'[42] contradicts the use of NT technology, because cellulosic ethanol production may require removal of residues from croplands. It is also debated whether food security and biodiversity are mutually exclusive goals[43]. These conflicts may be addressed through 'ecological intensification' based on the use of biological regulation to manage agroecosystems at field, farm and landscape scales[44]. In this context, natural ecosystems may provide examples of a number of interesting properties that could be incorporated into agroecosystems. Monitoring and assessment of sustainable land management as tools to promote sustainable use of soil resources are widely promoted by development organizations (World Bank and FAO) and bilateral programs (USAID)[45].

Soil degradation, i.e., a decline in the capacity of a soil to maintain ecosystem functions, is an important factor in evaluating ecosystem services and remains a major global issue[46]. Divergent views of agronomists and social scientists about the impact of degradation on ecosystem services must be resolved[47]. Apparent divergence in opinion and approaches is based on two contrasting views of increasing human population versus limited natural resources: (i) the Boserupian[48–50] view emphasizing the significance of human ingenuity in addressing ecological problems, and (ii) the Malthusian view reiterating that resources are finite and population increase can lead to degradation of natural resources. Drechsel et al.[51] surmised through analyses of data from 36 countries in sub-Saharan Africa that Malthusian mechanisms indeed are at work and are leading to soil nutrient depletion, accelerated erosion and reduced fallow periods caused by population pressure. This view illustrates the unsustainable population–agriculture–environment nexus on the African continent. The issue of severe decline in fertility of soils of Africa was also pointed out by Sanchez[52], as well as of global soil degradation by Bai et al.[46]. Should the

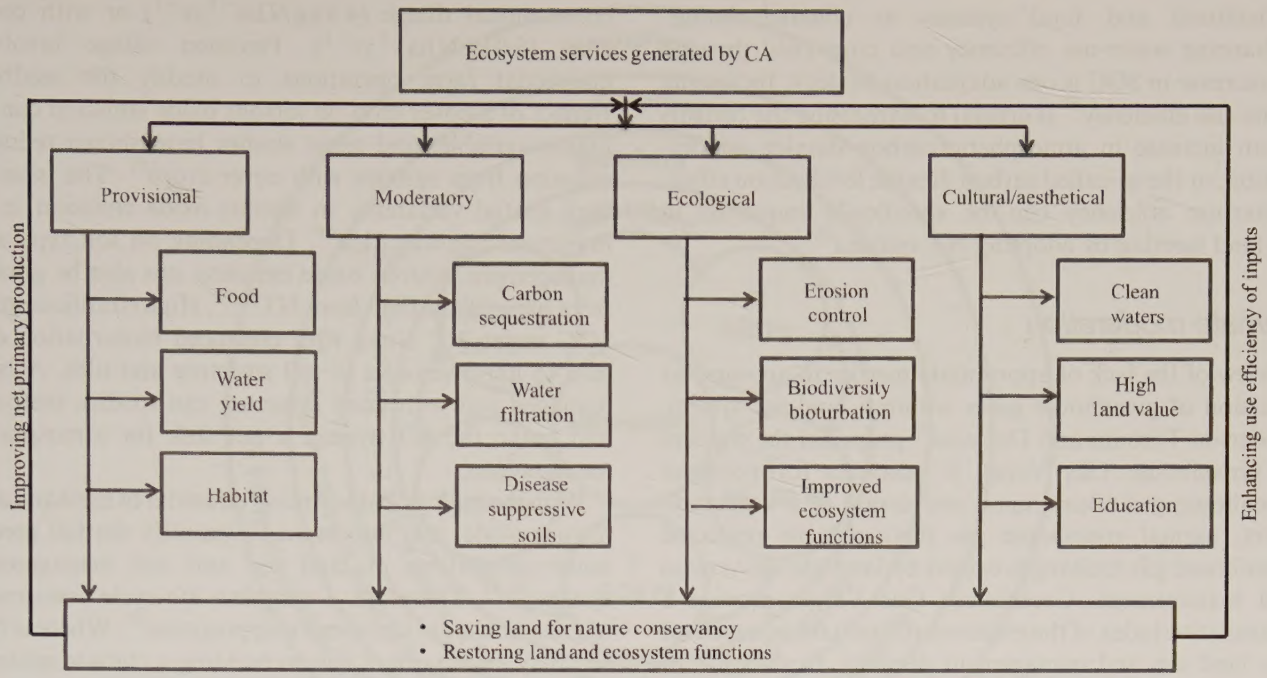

Figure 2. Range of ecosystem services generated by conservation agriculture (CA).

Boserupian view be considered sound, agricultural scientists and policy makers urgently need to identify and implement successful strategies to overcome current challenges. Difficulties encountered in implementing such strategies without effective community participation could exacerbate soil degradation and desertification[53]. Thus, Hurni[54] and Hurni et al.[55] proposed adoption of NT farming approaches with due consideration of social and economic dimensions of land-related issues.

Adoption of sustainable soil management technology can address these issues. Principal requirements of sustainable soil management are: (i) maintaining soil organic carbon (SOC) concentration at a level above the critical threshold in the root zone, (ii) optimizing soil physical quality including structure and water retention/transmission characteristics, (iii) managing plant nutrients, (iv) enhancing favorable soil biological properties, (v) improving root growth and proliferation, and (vi) reducing risks of soil erosion and other degradation processes[56]. Most of these requirements are met with a judicious and continuous use of NT for a long period (Fig. 2). Conversion to NT is especially useful in restoring/sustaining soil physical quality and reducing risks of water runoff and soil erosion, as indicated by soil bulk density, SOC dynamics, root growth, tilth, friability and hard-setting[57,58].

Ecosystem Services and Conservation Agriculture

By mimicking nature, discriminate and judicious implementation of NT can generate essential ecosystem services to meet specific demands of the growing human population, while also reducing risks of soil and environmental degradation. Principal ecosystem services provided and sustained through a long-term adoption of NT are discussed below.

Food security

Efficient use of inputs and agronomic productivity can be greatly enhanced by improving the concentration of SOC to optimal levels covering a wide range of crops and soils[59–63]. Enhancement of SOC to an optimum level increases crop yield by: (a) increasing plant available water capacity and nutrient supply (macro and micro) by promoting recycling and reducing losses, (b) restoring soil structure and tilth by bioturbation and microbial processes, and (c) decreasing soil erodibility and increasing water infiltrability, thereby reducing the risks of runoff and accelerated erosion. Despite notable improvements in soil quality, agronomic response to restoration of SOC in the root zone depends on a multitude of factors, including soil moisture and temperature regimes, availability of macro- and micro-nutrients, and various other soil properties[64,65]. Syntheses of global data from field experiments have shown a corresponding increase in yields of a variety of crops with an incremental increase in SOC in the root zone[64–66]. In general, low agronomic yields from degraded soils of sub-Saharan Africa, South Asia and other regions where SOC has been severely depleted by perpetual use of extractive farming are attributed to severe curtailment of beneficial rhizospheric processes in soil.

Improvement in the soil quality and resilience through restoration of SOC can also enhance adaptation of

agricultural and food systems to climate change. Enhancing water-use efficiency and crop yield through an increase in SOC is one adaptation strategy. Increasing water-use efficiency[67] is crucial to harnessing the benefits of an increase in atmospheric carbon dioxide concentration, or the so-called carbon dioxide fertilization effect. Water-use efficiency can be specifically improved in dryland farming by adopting NT systems[68].

Climate moderation

In view of the lack of appropriate metrics to account for emission of greenhouse gases within a land-use system, Anderson-Teixeira and DeLucia[69] proposed the concept of Greenhouse Gas Value. It accounts for potential greenhouse gas release upon conversion of a vegetation cover, annual greenhouse gas flux, and the projected greenhouse gas exchanges caused by land-use conversion and management. Greenhouse Gas Values provide a quantitative index of the emission-related consequences of any land use and management decision. Conversion of plow tillage to NT can enhance SOC concentration and stock[70–73] by reducing runoff, erosion and mineralization of soil organic matter. In South Africa, Mchunu et al.[74] observed that topsoil SOC stock was 26% higher under NT than under plowed soil, while NT reduced soil loss by 68% and SOC loss by 52%. However, the assessment of net gains in SOC must also consider the hidden C costs of inputs (such as fertilizers, pesticides, irrigation and other farm operations)[75]. In most cases, there is an increase in SOC concentration in the surface layer of soil managed by NT for a long time. However, there have been examples of lower concentration of SOC in the sub-soil layers of land managed with NT compared with that by plow tillage[76–78]. Determining differences in the accrual of SOC stock among tillage systems must be evaluated on a whole-profile basis rather than in the surface layer alone[77], and the experimental design must also use an adequate number of replications[79]. Over and above the effect of tillage methods and crop-residue management, SOC retention is also influenced by water management and irrigation system[80]. However, further increase in temperature under changing climate may decrease SOC[81], regardless of tillage system.

In general, adoption of NT on degraded agricultural lands has the potential to sequester SOC, while building fertility and restoring soil functions[82]. In addition to SOC (and carbon dioxide) dynamics, there are questions about other greenhouse gases (i.e., nitrous oxide and methane), which are also affected by tillage systems. In some soils, NT can lead to increased emission of nitrous oxide[83–85]. Irrespective of tillage, however, emission of nitrous oxide is highly variable and rates differ between arid and humid climates and fine- and coarse-textured soils. Smith et al.[86] observed that nitrous oxide emission from under corn (*Zea mays*) was the lowest from a precision-tillage treatment ($2.4\,kg\,N\,ha^{-1}\,yr^{-1}$) compared to conventional tillage ($4.9\,kg\,N\,ha^{-1}\,yr^{-1}$) or with cover crop ($5.0\,kg\,N\,ha^{-1}\,yr^{-1}$). Precision tillage involved additional farm operations to modify the seedbed. Impact of a cover crop on nitrous oxide emission can be highly variable, and some studies have shown reduced emission from systems with cover crops[87]. The issue of high spatial variability in nitrous oxide emission is an important consideration[88]. Depending on soil type and management, nitrous oxide emission can also be greater from plow tillage than from NT[89,90]. High stratification of SOC under NT along with enhanced bioturbation can lead to improvements in soil structure and tilth. A NT-managed soil with good structure can oxidize methane and make the NT system a net sink for atmospheric methane[91].

With the high global warming potential of methane and nitrous oxide, it is important to quantify the full greenhouse gas effects of land use and soil management systems[69,92]. Therefore, a complete life cycle assessment of greenhouse gas emissions is appropriate[93]. Whereas NT can play an important role in building a climate-resilient farm, the role of associated components (i.e., water, energy, fertilizers, residues and cover crops) must also be considered[94]. Residue management (burning/removal or incorporation) can also impact the greenhouse gas budget of a farming/cropping system[93].

While conservation management of world soils has a large technical potential to sequester SOC and offset anthropogenic emissions[95], there are numerous challenges to realizing this potential[96]. In general, challenges posed by humans (e.g., land rights, social equity and cultural traditions) are often more severe than those posed by biophysical constraints (e.g., soil type, climate, terrain and slope gradient).

Biodiversity

Soil is an important habitat for biota. Soil organisms, comprising an extremely diverse and complex biological community, are affected by land use and soil management systems[97]. A strong link exists between biodiversity and human wellbeing[98]. Thus, habitat needs for other species must not be jeopardized in a managed ecosystem[99]. Soil biota generate numerous ecosystem services, such as elemental cycling, denaturing of pollutants, soil aggregation, and insect and disease control[100–107]. While agricultural intensification generally decreases biodiversity[108], adoption of NT enhances biodiversity and strengthens soil functions and ecosystem services[109]. In strong contrast to NT, plow tillage can diminish microbial biomass and enzyme activity[110]. Thus, to enhance biodiversity, a NT farming system will be better than one with plow tillage. In general, NT has the potential to diversify the suite of ecosystem services through judicious management[111], notably less soil disturbance. Specific indicators of biodiversity are needed for croplands in diverse biomes[112]. It may be appropriate to assess biodiversity

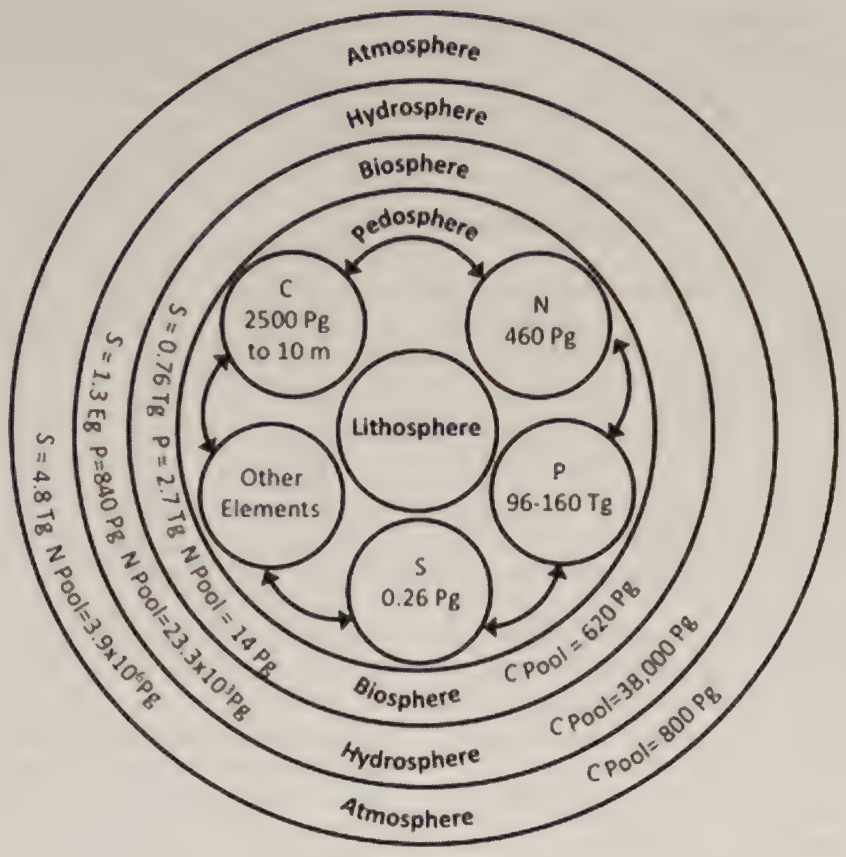

Figure 3. Schematic of the coupled cycling of elements within the pedosphere. Fluxes of elements in the pedosphere with those in the biosphere, hydrosphere and atmosphere are well known[115]. Data on carbon pools are from Lal[91]. Data on nitrogen pools are from Robarts and Wetzel[116] and those of P and S are from Stevenson[117].

in multi-functional landscapes[113], and species identity may be even more important than species diversity[114].

Elemental cycling

The earth/soil has numerous elements with their own cycles, either singularly or coupled. Coupled cycling of elements is the rule, rather than an exception. Major elements (carbon, nitrogen, phosphorus and sulfur) have coupled cycling within the pedosphere, and among the pedosphere, lithosphere, atmosphere, hydrosphere and biosphere through an inter-connected web of global cycles (Fig. 3)[91,115–117]. By conserving carbon and water, NT strengthens the coupled cycling of numerous elements. Coupled cycling of carbon and water at a global scale indicates the inter-dependence of these elements (Fig. 4)[91,95,118–121]. Within a landscape, long-term use of NT strengthens elemental cycling. For example, NT stratifies SOC by concentrating it within the surface layer, conserving water by reducing runoff and evaporation, increasing water storage in the root zone, increasing plant-available water capacity, and increasing net primary production by reducing risks of drought and decreasing losses of plant nutrients by runoff, leaching and erosion. The return of crop residues to soil, an integral component of NT, is crucial to elemental cycling and reducing the requirement for nutrient replenishment with chemical fertilizers.

Soil erosion, sedimentation and non-point source pollution

Surface mulch on unplowed and consolidated soil decreases erodibility, increases resilience and drastically reduces erosivity (e.g., rain drop impact, shearing force of overland flow and blowing wind, and wind-driven rain). Data in Table 2 from the North Appalachian Experimental Watersheds at Coshocton, Ohio, indicate the relative effectiveness of NT farming in erosion control, especially for some exceptional rainstorm events[122]. Data in Table 3 from western Nigeria also show the effectiveness of NT in reducing runoff and soil erosion up to a slope gradient of 15% in an environment characterized by highly erosive tropical rains[123].

Despite the conservation effectiveness of NT, it does have some disservices. Surface applications of fertilizers and pesticides/herbicides can accentuate the risk of transport in runoff, especially during the first few events after application[124–126]. In Ohio, risk of herbicide transport in water runoff was high for an erosive rainfall event that occurred within 10 days of application, and detectable levels could be observed 50–60 days after application. Shipitalo et al.[127] observed concentration of atrazine and alachlor in runoff high enough to be of concern even for NT, and concentration of herbicides in the first few runoff events after application can be well

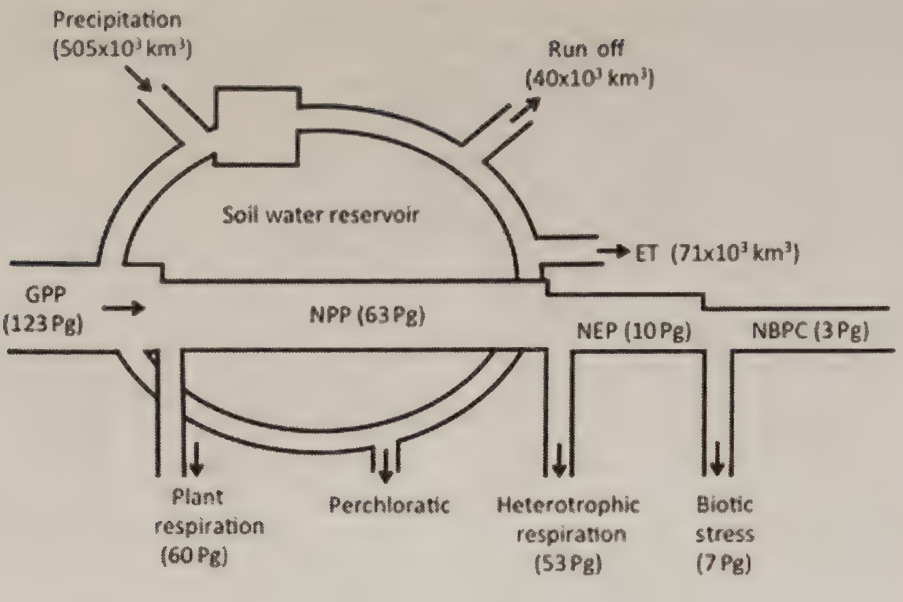

Figure 4. Schematic of the coupled cycling of carbon and water. Data for C fluxes are from multiple sources[91,95,118,119] and those for water are from Ogle et al.[120] and Bengtsson[121].

Table 2. No-till effects on soil erosion from cropland watersheds at Coshocton, Ohio[116].

Tillage system	Length of record (years)	Soil erosion ($Mg\,ha^{-1}\,yr^{-1}$)	Ratio
Conventional (prevailing)	28	11.57	578.5
Improved	22	3.23	161.5
Minimum tillage	10	0.87	43.5
No-tillage	5	0.02	1

Prevailing: straight/sloping rows, low fertility.
Improved: contour rows, moderately high fertility.
Minimum tillage: plow and plant.
No-tillage: chemical weed control, corn stover mulch, manure.

above drinking water standards. In addition, flow of pesticide-containing water through macropores and preferential pathways can be an issue with NT systems[128]. If an intense rainstorm were to occur soon after surface application of chemicals to soil with well-developed macroporosity, water transmitted into tiles and shallow ground water by macropores could contain significant concentration of chemicals[129].

Conservation Agriculture and Relationship among Multiple Ecosystem Services

Strong links exist among diverse ecosystem services, but managing exclusively for one (i.e., food production) can lead to a substantial decline in another (i.e., biodiversity). Therefore, it is important to understand theoretical relationships among ecosystem services. Bennett et al.[130] proposed three sets of issues to understand relationships among ecosystem services: (i) nature of ecosystem service (i.e., flow across a landscape or specific processes that regulate the nature of the relationship), (ii) trade-offs and synergism (i.e., effective ways to mitigate trade-offs or change of relationship among ecosystem services over time, with management and across scales), and (iii) regulating services and regime shift (i.e., nature of shifts or variation in ecosystem services with decline in regulating services). Trade-offs may occur among ecosystem services, and specific disservices in agroecosystems must be carefully evaluated at appropriate temporal and spatial scales, since the negative impacts are difficult to reverse (i.e., erosion, non-point source pollution)[35]. Trading ecosystem carbon storage for food[131] has important managerial implications that require an objective and careful evaluation. Thus, agricultural ecosystems must be appropriately managed to optimize diversity, composition and functioning of remaining natural ecosystems in the landscape[132]. In some cases, adoption of organic agriculture may strengthen specific ecosystem services such as pollination, biological control and nutrient cycling[133]. Yet, low productivity of organic systems is an issue in the world with an ever increasing population and escalating demand for ecosystem services.

Payments for Ecosystem Services

Innovative agricultural systems must be developed to maximize ecosystem function and services and minimize disservices. Therefore, linkages and inter-dependence between food production and other ecosystem services for recommended management practices must be critically assessed. Adoption of recommended management practices by land managers, especially resource-poor

Table 3. Mulching effects on surface runoff and soil erosion from an Alfisol in western Nigeria from April to August 1973 (recalculated from Lal[123]).

	Runoff (mm/season)					Erosion (Mg ha^{-1})				
Slope gradient (%)	1	2	3	4	5	1	2	3	4	5
1	316	0	55	1	20	7.5	0	1.2	0	0.6
5	347	7	159	2	81	80.4	0	8.2	0.2	5.6
10	311	20	52	20	51	152.9	0.1	4.4	0.1	3.2
15	317	17	90	21	46	155.3	0	23.6	0.1	7.6
Mean	323	11	89	11	50	99.0	0.025	9.4	0.1	4.25
Ratio	29	1	8	1	5	3960	1	376	4	170

Rainfall = 781 mm.
1 = Plowed fallow, 2 = corn, plowed, mulched, 3 = corn, plowed, 4 = corn, no-till, 5 = cowpea, plowed.

farmers, can be promoted through payments for ecosystem services (i.e., carbon sequestration, water resources, improvement and biodiversity enhancement). Establishment of a rational price (i.e., carbon credits or green water credits) may require identification of soil/site-specific indicators[134] at different spatial scales and with hierarchical perspective. These indicators can provide the basis of an appropriate framework. High negative external costs associated with food production[135] could be minimized through the use of appropriate indicators. One option would be to determine suitability of a provision by collective management or cooperative solution based on resource characteristics[136]. Another option would be to reduce disservices and negative costs through payments for enhancing ecosystem services and reducing disservices. A third option would be a market-based approach[137].

Market-based approaches could reduce risks of the 'tragedy of the commons'. The strategy would be to harness market potential to provide ecosystem service incentives to land managers. Markets could be used as a tool for organizing the supply of ecosystem services to and from agroecosystems[137]. Being myopic by nature, humans' desires to make quick economic gains by cutting corners could degrade soils and jeopardize natural resources. Thus, assessing societal value of ecosystem services would be relevant, especially in developing countries[138] with predominantly resource-poor farmers and smallholders. A just pricing system for ecosystem services could promote the adoption of NT and recommended management practices.

A market-based tool could be used either as: (i) economic incentive or (ii) performance payment[139]. The latter, paid in cash or kind, could be made conditional on achieving a well-defined action or outcome. For example, carbon credits could be sold only if farmers were to adopt and maintain NT for a specific duration of time (e.g., 6–10 years), and payments could be made at the end of the designated period. Determining the rational value of a specific ecosystem service would be critical to the success of the program. Undervaluing a service (such as SOC sequestration) could lead to its misuse and depletion. No one protects or safeguards an under-valued resource. The market-based approach for voluntary carbon trading was not successful, because of the unfair price governed by the lack of cap on fossil fuel combustion. However, payments to farmers for providing essential ecosystem services (e.g., adapting and mitigating abrupt climate change, improving water quality and its renewable supply, and enhancing biodiversity) based on a fair and just price would incentivize land managers toward adoption of sustainable technologies.

Constraints to Adoption of No-till

Despite numerous advantages, cropland area under NT is hardly 9% worldwide, mostly in countries where large-scale commercial farming is practiced (e.g., USA, Canada, Brazil, Argentina, Chile, Paraguay and Australia). Adoption of NT is practically negligible by resource-poor, smallholders of sub-Saharan Africa, South and South-East Asia, Central America, Caribbean and Pacific Islands. Yet, these are also the regions where the potential benefits of NT are high[140]. Reasons for low adoption of NT vary among developing and developed economies, climatic regions, soil types and farming systems. In developing economies, farmers may not have access to herbicides and appropriate seeding equipment. Competing uses of crop residues for fodder and fuel may be another constraint. Land ownership and tenure, important to investment in long-term improvement in soil restoration, is another issue that limits adoption. Furthermore, NT farming may require more skills and experience in its implementation.

In temperate regions, spring-time soil temperature is often sub-optimal and seedling emergence and crop establishment are slower under NT compared with plow-based seedbed preparation. Thus, crop yield can be depressed under colder, wetter climates[120,141]. Nonetheless, intensified rotations with NT in the US Great Plains can be more profitable than plow-based systems, even when grain yields are slightly lower[142]. However, even a slight yield decrease may be perceived as risky to

farmers. There is also a range of socio-cultural/economic reasons for low adoption of NT[143].

Conclusions

Ecosystem services providing benefits to the environment must be sustained to meet the increasing and diverse demands of humanity. Soil provides all four categories of ecosystem services (e.g., provisioning, moderating, ecological and cultural/aesthetical) and its resource must be sustainably used and restored, because it is a finite resource and prone to degradation and depletion. Sustainable management of soil involves: (i) replacing what is removed, (ii) restoring what has been degraded, and (iii) responding prudently to off-site effects of management-induced alterations in ecosystem services (e.g. greenhouse gas emission, non-point source pollution and sedimentation) and disservices.

Adoption of NT has its own potential and challenges. Despite more than 50 years of research in temperate and tropical climates, NT farming is adopted on only 124.8 Mha (~ 9% of cropland area). Frustratingly, adoption of NT has been especially low with resource-poor farmers and smallholders in developing countries, where it would otherwise be desperately useful.

Among numerous constraints to the adoption of NT is the competing use of crop residues harvested for cattle feed, cooking fuel, construction, and other domestic and commercial uses. Thus, a market-based approach is needed to promote adoption of NT through economic incentives or performance payments for ecosystem services, which could also reduce disservices or trade-offs. To be effective, however, rational value of a specific ecosystem service (i.e., SOC sequestration) must be objectively assessed. Undervaluing an ecosystem service would jeopardize the finite soil resource and aggravate disservices. No-till farming, conservation agriculture and sustainable land management are proven innovative options that maintain or enhance ecosystem services and reduce risks of disservices. Adoption of NT should be expanded to sustain or improve ecosystem services to meet the needs of a growing world population.

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Using biodiversity to link agricultural productivity with environmental quality: Results from three field experiments in Iowa

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Abstract

Agriculture in the US Corn Belt is under increasing pressure to produce greater quantities of food, feed and fuel, while better protecting environmental quality. Key environmental problems in this region include water contamination by nutrients and herbicides emitted from cropland, a lack of non-agricultural habitat to support diverse communities of native plants and animals, and a high level of dependence on petrochemical energy in the dominant cropping systems. In addition, projected changes in climate for this region, which include increases in the proportion of precipitation coming from extreme events could make soil and water conservation in existing cropping systems more difficult. To address these challenges we have conducted three cropping systems projects in central Iowa: the Marsden Farm Cropping Systems experiment, the Science-based Trials of Row-crops Integrated with Prairies (STRIPs) experiment, and the Comparison of Biofuel Systems (COBS) experiment. Results from these experiments indicate that (1) diversification of the dominant corn–soybean rotation with small grains and forage legumes can permit substantial reductions in agrichemical and fossil hydrocarbon use without compromising yields or profitability; (2) conversion of small amounts of cropland to prairie buffer strips can provide disproportionately large improvements in soil and water conservation, nutrient retention, and densities of native plants and birds; and (3) native perennial species can generate large amounts of biofuel feedstocks and offer environmental benefits relative to corn- and soybean-based systems, including greater carbon inputs to soil and large reductions in nitrogen emissions to drainage water. Increasing biodiversity through the strategic integration of perennial plant species can be a viable strategy for reducing reliance on purchased inputs and for increasing agroecosystem health and resilience in the US Corn Belt.

Key words: biodiversity, multifunctionality, conservation, cropping systems

Introduction

The US Corn Belt, spanning from western Ohio to eastern Nebraska and from southern Minnesota to northern Missouri, comprises one of the largest, most productive rain-fed grain regions in the world. Its natural endowment with fertile soils and a favorable climate makes it a major focal point in discussions of how to supply an expanding world population with enough food and plant-derived renewable energy. These discussions increasingly focus not only on the technologies of farming but also on the environmental impacts of farming practices and the resilience of agricultural systems to changes in climate, energy and nutrient availability, and other factors acting at a global scale.

Patterns of agricultural development, land use and environmental quality in Iowa exemplify the directions that conventional agriculture has followed in the Corn Belt during the past half-century. Iowa is now the leading US producer of corn, soybean, hogs, eggs and ethanol, and ranks seventh in cattle production[1,2]. Concomitantly, the state also ranks high nationally in the number of surface waters impaired by excessive concentrations of nutrients, pathogens, pesticides and soil sediment, and its croplands are major sources of nitrogen and phosphorus delivered to the hypoxic zone in the Gulf of Mexico[3–5].

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In addition to high levels of water pollution, the development of conventional agricultural systems in the Corn Belt has been accompanied by a major loss of biological diversity. Tall-grass prairie was the dominant land cover in Iowa and much of the central and western Corn Belt prior to Euro-American settlement[6]; smaller areas within the region were occupied by other types of perennial plant communities, including savannas, riparian forests and wetlands. Most of this native vegetation is now gone, having been converted to cropland or pasture by 1900. At present, Iowa ranks last nationally in the amount of pre-settlement vegetation still remaining[7], with the state's prairie communities reduced to <0.1% of their former area[8]. Change has also taken place in recent years where land that had been in perennial cover through enrollment in the Conservation Reserve Program (CRP) has been converted to row crop production. Schilling and Spooner[9] reported that in Iowa's Squaw Creek watershed, 9% of the area that had been in CRP grassland in 1990 was converted to corn and soybean production by 2005. Not surprisingly, given the loss of native plant communities and conservation grasslands, the diversity of Iowa's native flora and fauna has been greatly reduced[10].

A substantial loss of crop diversity has also occurred throughout the central Corn Belt, illustrated by historical changes in land use in Iowa. Although the total amount of cropland in Iowa has remained relatively constant over the past 60 years, there has been a marked shift toward corn and soybean production; these two crops now occupy two-thirds of the state's total land area[11]. Between 1949 and 1997 in Iowa's Raccoon River watershed, the percentage of cropland used for wheat, barley, oat, alfalfa and other hay crops fell from 42 to 3%; in contrast, the proportion in corn and soybean grew from 57 to 97%[12]. Historical changes in crop diversity in the Corn Belt are linked with changes in livestock production in the region. From 1945 to 2000, cattle numbers in the Corn Belt declined by 52% as beef production shifted westward; concomitantly, land used for hay and oat production, used to feed cattle and other livestock, declined by 60 and 97%, respectively[13]. Similarly, in an analysis of landscape change during 1937–2002 in three Iowa townships, Brown and Schulte[14] noted marked reductions in the area of grasslands suitable for pasturing cattle.

Reductions in crop and non-crop diversity have been linked to a variety of forms of environmental degradation and agroecosystem dysfunction; conversely, increases in diversity can be associated with improvements in agroecosystem health and function. For example, Hatfield *et al.*[12] reported that nitrate concentrations in Iowa's Raccoon River have increased since 1970 as land use in adjacent cropland has shifted from diverse cropping systems containing small grains, hay, corn and soybean to simpler corn–soybean systems, which are more prone to nutrient leaching. Reductions in diversity and accompanying reductions in ecosystem functioning have also occurred due to removal of non-crop vegetation. In pursuit of increased field size and greater ease in the use of large agricultural machinery, trees on field borders have been removed in many areas of the Midwestern USA. Yet, fields that lack adjacent shelterbelt trees can be subject to increased wind speeds and greater wind erosion, and have reduced crop water use efficiency and lower yields[15,16]. Non-crop vegetation can also affect pest dynamics. In a comparison of Midwestern US landscapes dominated by corn and soybean fields versus landscapes containing a mix of crop fields and non-crop habitat, Gardiner *et al.*[17] found that soybean aphid (*Aphis glycines*) was subject to less biological control by natural enemies and consequently was more problematic in the less diverse landscapes, whereas greater diversity at a landscape level enhanced predator populations and biological control of the pest.

Given the loss in the Corn Belt of native biodiversity, crop diversity and the ecosystem services that diversity can provide, we ask: Are impoverished biological communities and environmental degradation unavoidable costs of high agricultural productivity or can farming systems be developed to provide both high yields and necessary ecosystem services, including water quality protection, soil conservation and pest control? Schulte *et al.*[18] suggested that the strategic integration of perennial vegetation into agricultural landscapes dominated by annual crops could help reconcile agricultural productivity with environmental quality. Increasing crop and non-crop diversity in agroecosystems with perennial species may be especially important because perennial plants perform many key ecological functions more effectively than annuals, including (1) regulating the flow and storage of water, (2) reducing soil erosion, (3) storing and cycling nutrients and carbon, and (4) providing habitat for natural enemies of crop pests and for native plants and animals of conservation concern[18,19].

Many options exist for diversifying agroecosystems with perennial vegetation, including forage crops that are alternated with row crops in rotation sequences; riparian buffers of trees, shrubs and grasses that separate crop fields from streams; in-field strips of herbaceous species that filter runoff water; trees and shrubs on field margins that reduce wind movement; reconstructed wetlands that receive drainage water from crop fields; agroforestry plantations that supply a variety of economic products; and woody and herbaceous plants that are harvested as biofuel feedstocks[18]. Here, we describe some of the agronomic, economic and environmental consequences of three approaches for increasing diversity in agroecosystems in Iowa: adding perennial legumes to corn–soybean rotations; weaving strips of prairie vegetation into corn and soybean fields to serve as filter strips; and growing prairie vegetation to produce solid or liquid fuel.

Lengthening Rotations to Reduce Agrichemical Use While Maintaining Productivity: The Marsden Farm Cropping Systems Experiment

Conventional corn and soybean production is a chemically intensive activity. Together, corn and soybean receive more herbicide active ingredients and more synthetic fertilizer than any other crops grown in the USA[20,21]. Emissions of herbicides and nitrogen from fields to water in the Corn Belt are roughly proportional to the quantities of those materials applied and the percentage of a watershed occupied by corn and soybean[22–25]. In the case of herbicides, emissions include both runoff and leaching losses; for nitrogen, emissions result from runoff and leaching of fertilizer, as well as losses from mineralized soil organic matter and manure[26]. Unlike deep-rooted perennial forage crops whose root systems are active for many months each year, corn and soybean have relatively shallow root systems that are active for only part of the year, rendering those crops leaky with regard to nitrogen loss during the spring and fall, when rainfall can be abundant and can transport nitrogen through the soil[12,27].

Concentrations of herbicides and nitrate in surface and ground water in the Corn Belt can exceed health-based standards for drinking water quality, raising concerns for impacts on human health, wildlife and fisheries[3,28–30]. Magdoff *et al.*[31] and Dinnes *et al.*[32] noted that reductions in synthetic nitrogen fertilizer use, coupled with greater use of perennial legume crops and judicious use of manure, constitute important components of strategies to conserve nitrogen and protect water from nitrogen contamination. Similarly, reducing herbicide use by increasing reliance on physical weed control practices and diverse crop sequences that challenge weeds with a broad range of stress and mortality factors may be an effective approach for reducing herbicide concentrations in the hydrologic system[3,33,34].

To test the hypothesis that diversifying a simple corn–soybean cropping system could allow for substantial reductions in nitrogen fertilizer and herbicide use without compromising crop productivity and profitability, a 9-ha field experiment was established in 2001 at the Iowa State University Marsden Farm, in Boone Co., IA. The site lies within a region of intensive rain-fed corn and soybean production and is surrounded by farms with high levels of productivity. Soils at the site are fertile Mollisols. In addition to a conventionally managed 2-year rotation (corn–soybean) that receives fertilizers and herbicides at rates comparable to those used on surrounding commercial farms, the experiment includes two more diverse low-external-input (LEI) cropping systems: a 3-year rotation (corn–soybean–small grain + red clover) and a 4-year rotation (corn–soybean–small grain + alfalfa–alfalfa) managed with reduced N fertilizer and herbicide inputs. The 2-year rotation is typical of cash grain farming systems in the region, whereas the 3-year and 4-year rotations are representative of LEI farming systems in the region that include livestock.

The experiment used a randomized complete block design with each crop phase of each rotation system present every year in four replicate blocks. Plots were 0.15 ha and managed with conventional farm machinery. After uniformly cropping the site with oat in 2001 and tuning the three management systems in 2002, intensive data collection began in 2003. Spring triticale was used as the small grain in 2003–2005, whereas oat was used in 2006–2010. During 2003–2010, solid pack manure was applied during the fall preceding corn production in the 3- and 4-year rotations at a mean dry matter rate of 9 Mg ha^{-1}, providing a mean of 120 kg total N ha^{-1}. Corn in the 2-year rotation received 112 kg N ha^{-1} as urea at planting, whereas corn in the 3- and 4-year rotations did not. The late spring nitrate test[35] was used to determine rates for post-emergence side-dress N applications (as urea ammonium nitrate) for corn in all rotation systems. Weed management in the 2-year rotation was based largely on herbicides applied at conventional rates. In the 3- and 4-year systems, herbicides were applied in 38-cm-wide bands over corn and soybean rows rather than broadcast, greater reliance was placed on cultivation, and no herbicides were applied in small grain and forage legume crops. Choices of herbicides used in each system were based on the identities, densities and sizes of weed species observed in the plots. Sampling procedures and other details of farming practices used in the different cropping systems during 2003–2010 are described in Liebman *et al.*[36], Cruse *et al.*[37] and Gómez *et al.*[38].

Results of the experiment indicated that corn and soybean yields from the more diverse 3- and 4-year systems were higher than those from the conventional system, despite substantial reductions in the use of synthetic N fertilizer and herbicides[36–38] (Table 1). Fossil energy use decreased with increases in crop diversity due to reductions in the use of fertilizers and pesticides, gas to dry grain, and liquid fuel to power farm machinery[37] (Table 1). Weed suppression was highly effective in each of the three cropping systems as reflected by the very small amounts of weed biomass found in corn and soybean phases of the rotations[36] (Table 1). Although labor requirements were greater in the longer, more diverse rotation systems, net returns to land and management were statistically equivalent in all three cropping systems, with a slight numerical advantage for the 3- and 4-year systems (Table 1).

Biological diversity contributed to the successful functioning of the LEI systems used at the Marsden Farm in multiple ways. For example, though triticale and oat added relatively little revenue themselves to the 3- and 4-year systems[36], they served as effective nurse crops for establishing red clover and alfalfa, thereby minimizing

Table 1. Inputs, yields, weed biomass and net returns for the three cropping systems in the Marsden Farm rotation experiment, Boone Co., IA, 2003–2010. Fossil energy inputs are yearly means for 2003–2008; all other data are yearly means for 2003–2010. Within rows, means followed by different letters are significantly different ($P < 0.05$); means not followed by letters are statistically equivalent. Data are from Liebman *et al.*[36], Cruse *et al.*[37] and Gómez *et al.*[38].

	Cropping system		
	2-year rotation: Corn–soybean	**3-year rotation: Corn–soybean–small grain[1]/red clover**	**4-year rotation: Corn–soybean–small grain[1]/alfalfa–alfalfa**
Whole rotation			
Fertilizer N inputs, kg N ha^{-1} yr^{-1}	76 a	18 b	12 c
Herbicide inputs, kg active ingredients ha^{-1} yr^{-1}	1.91 a	0.28 b	0.21 c
Fossil energy inputs, GJ ha^{-1} yr^{-1}	9.1 a	4.9 b	4.3 c
Labor requirements, h ha^{-1} yr^{-1}	1.8 c	2.8 b	3.5 a
Net returns to land and management[2], $ ha^{-1} yr^{-1}	690	705	701
Crop yields			
Corn, Mg ha^{-1}	12.2 b	12.6 a	12.8 a
Soybean, Mg ha^{-1}	3.4 b	3.7 a	3.8 a
Small grain[1], Mg ha^{-1}	–	3.7	3.8
Alfalfa, Mg ha^{-1}	–	–	9.0
Weed biomass			
In corn, kg ha^{-1}	2.0	4.6	3.7
In soybean, kg ha^{-1}	1.5	3.8	3.6

[1] Triticale was grown as the small grain crop in 2003–2005; oat was used in 2006–2010.
[2] Crop subsidy payments were not included as sources of revenue.

erosion and reducing weed growth in the absence of herbicides.

Although forage legumes were less profitable than corn[36], their inclusion within the diversified rotation systems allowed substantial reductions in nitrogen fertilizer use for corn production[39,40] (Table 1). Harvest of alfalfa hay and concentrated feeds allowed integration with livestock production and fostered nutrient return to cropland through the application of manure[36], further reducing production costs and fossil energy consumption[37]. In studies conducted by Randall *et al.*[27] and Drinkwater *et al.*[41], leaching losses of nitrogen from deep-rooted perennial forage crops, and from corn–soybean systems diversified with forage legumes and small grains were lower than from corn and soybean. Although we did not measure nitrogen losses in our study, those results suggest that nitrogen losses from the diverse LEI systems in the Marsden Farm experiment are likely to have been lower than from the simpler conventional system.

As in our study, other research has shown that lengthening rotations beyond 2 years can increase soybean yields[42,43]. In addition, both corn and soybean yields have been shown to respond positively to manure application even when the nutrient requirements of those crops are met with other fertility sources[44–46]. In 2010, the incidence and severity of sudden death syndrome of soybean, caused by the pathogen *Fusarium virguliforme*, and subsequent losses of soybean yield were much lower in the 3- and 4-year rotation systems than in the 2-year rotation, though the mechanisms for these effects are not yet clear (L.F. Leandro, pers. comm.).

Finally, diversifying the corn–soybean system with small grain and forage crops increased the diversity of habitats available to insects and rodents that preyed upon weed seeds[47–49], which is likely to have stabilized seed predator populations and increased their effectiveness in suppressing weed population growth under conditions of reduced herbicide inputs[50].

The rotation systems and management practices used in the Marsden Farm experiment are well suited to investigations of crop performance, weed dynamics, and economic and energetic costs and returns, but they should not be construed to represent optimal systems. The experiment has not addressed the need for market development for diverse crops to be viable at a larger scale, nor has the experiment provided direct data concerning the impact of the contrasting cropping systems on nutrient and pesticide emissions. Nonetheless, results from the experiment indicate diversified LEI systems for agronomic crops in the US Corn Belt can be highly productive and economically viable, and that further research into their refinement and environmental impacts is warranted. Research focused on the social and economic conditions that help to maintain the dominance of simple cropping systems and that discourage farmers from exploring and adopting more diverse LEI systems would be a useful complement to investigations targeting biophysical characteristics of contrasting cropping systems.

Using Native Perennial Vegetation to Improve Conservation in Annual Cropland: The Science-based Trials of Row-crops Integrated with Prairies (STRIPs) Experiment

Soil and water conservation are critical components of any cropping system, yet, existing rates of erosion within the US Corn Belt indicate that despite high levels of crop productivity, substantial conservation challenges need to be addressed[51,52]. Conservation practices that have proven useful in restricting the movement of soil sediment within and out of crop fields include buffers and filter strips composed of herbaceous plants, typically perennial grasses. An extensive review by Dosskey *et al.*[53] found that sediment trapping efficiencies of buffers generally range between 40 and 85% for buffers that are as small as 2.5% of the source area. Modeling results predict a relatively rapid increase in sediment trapping efficiency as the percentage of buffer area to source area increases from 0 to 10%, with trapping efficiency leveling off as the percentage increases above 10%[53]. Distributing perennial cover across several contour strips within a watershed in addition to buffers at the bases of catchments may enhance sediment and runoff trapping efficiency since most sediment deposition occurs in the first few meters of the up-slope edge of buffers[54].

In addition to the amount of vegetation cover, the composition of herbaceous vegetation used for conservation buffers can affect their function. Cool season exotic grasses, such as smooth brome (*Bromus inermis*) and Kentucky bluegrass (*Poa pratensis*), are widely used to provide ground cover in agricultural areas of the US Corn Belt, but are relatively weak-stemmed and prone to lying flat under heavy rain. In contrast, native tallgrass prairie communities are typically dominated by stiff-stemmed warm season grasses, such as Indiangrass (*Sorghastrum nutans*), big bluestem (*Andropogon gerardii*) and switchgrass (*Panicum virgatum*), and a wide range of erect forbs, including species in the genera *Solidago*, *Aster*, *Rudbeckia*, *Ratibida*, *Silphium*, *Helianthus* and *Monarda*[55], which are less prone to collapse under heavy rain, and which are more effective in providing resistance to water flow and sediment movement[56,57].

In addition to reducing water runoff and soil erosion from crop fields, strips and patches of perennial native plants in and around agricultural areas can provide habitat to support native animal populations, including species of conservation concern[58–63]. Many bird species respond positively to agricultural conservation practices such as grassed waterways[64,65], field borders[66,67], and riparian buffer strips[62,68]; the general trend among these studies is increased bird presence, abundance and richness where those small habitats are present adjacent to row crops. Compared with monocultures of cool-season grasses, diverse prairie communities are expected to provide higher quality bird habitat and harbor more bird species[65,69,70].

Beginning in 2004, 12 experimental watersheds (0.5–3.2 ha each) were established within the Neal Smith National Wildlife Refuge (NSNWR), in Jasper Co., IA, to test the hydrologic and biodiversity effects of integrating prairie conservation buffers with row-crop production. The four treatments in the experiment consisted of varying proportions and different configurations of reconstructed native prairie species within corn and soybean fields. For the 100% crop treatment, each replicate watershed was used entirely for corn and soybean production, with the different crops grown in alternate years. There were two 10% prairie/90% crop treatments, one with prairie vegetation concentrated at the toe slope position and a second in which prairie vegetation is distributed in contour strips along the slope, including at the toe slope. In addition, there was a 20% prairie/80% crop treatment that also had prairie vegetation distributed in contour strips and at the toe slope. The experiment used a balanced incomplete block design with three replicates of the four treatments in four blocks. Soils in the experimental watersheds are Alfisols and Mollisols. Analysis of pretreatment data indicated that the watersheds had similar slopes, soil textures, and soil C and N concentrations.

Prior to the start of the experiment, exotic cool-season forage grasses, primarily smooth brome, dominated each of the watersheds. After tillage in the fall of 2006 to kill sod and level the soil, crop production in all watersheds was conducted using continuous no-tillage techniques; soil disturbance occurred only during corn and soybean planting and injection of anhydrous ammonia for corn production. Soybean was grown in 2007 and 2009 and corn was grown in 2008 and 2010 using conventional fertilizer and herbicide practices. Native prairie species were seeded in toe slope and strip positions in July 2007. The strips of prairie species had a minimum width of 4 m and were spaced at a minimum distance of 36 m to accommodate agricultural machinery operations.

Each watershed had a distinct surface flow outlet point where an H-flume was installed to monitor volume and rate of surface runoff. Each of these locations was also instrumented with an automated water sampler to obtain flow measurements and discrete water samples based on flow intervals. In addition, belowground water quality was monitored with a network of suction lysimeters and groundwater wells. Surface and ground water measurements were conducted in 2007–2010. Vegetation surveys were conducted between July and August in 2008, 2009 and 2010 to estimate species richness and determine plant cover using sampling quadrats along transects within the prairie and cropped areas of the watersheds. From 2007 to 2010, all watersheds were visited 9–12 times during spring and summer months and surveyed for birds using spot

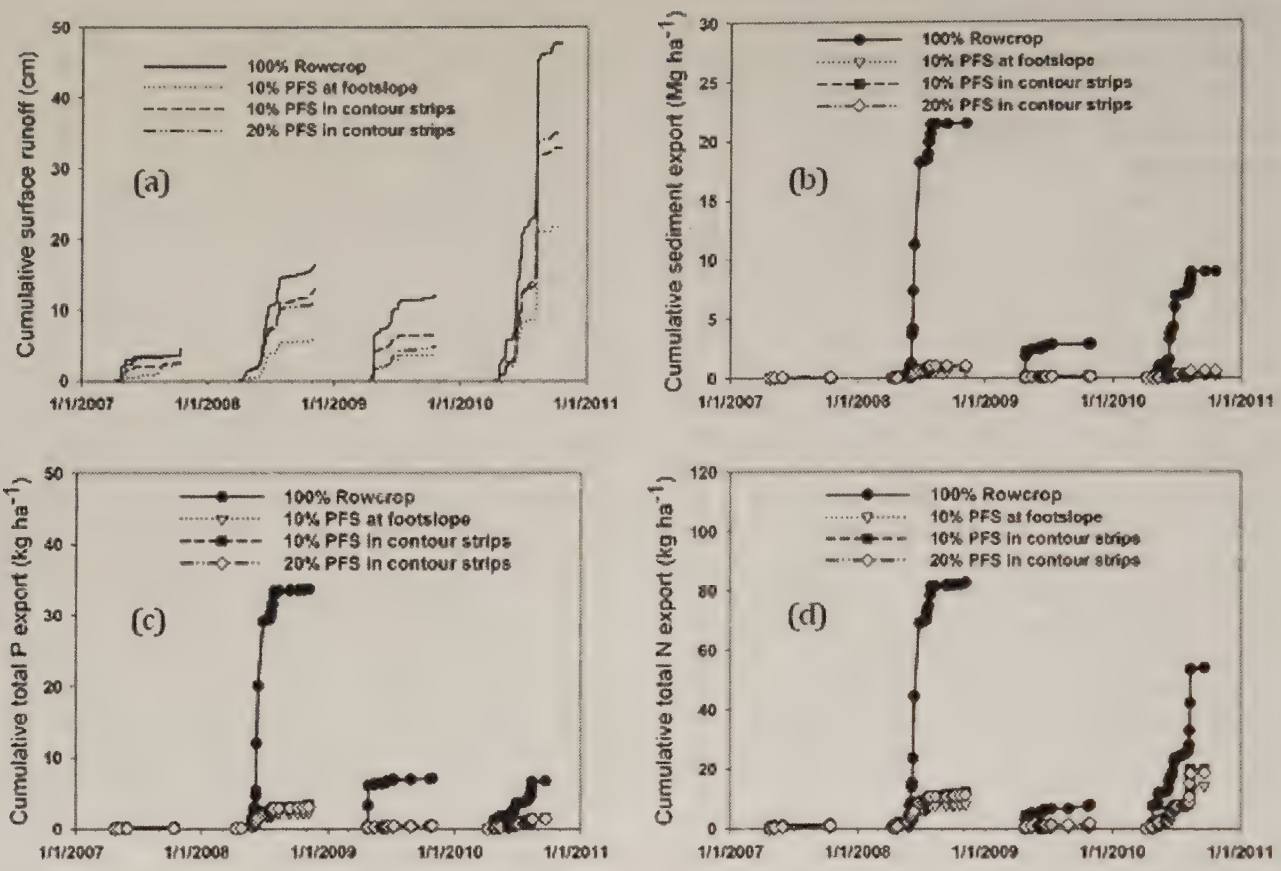

Figure 1. Cumulative (a) surface runoff[72], (b) export of sediment[73], (c) total phosphorus export (Helmers *et al.*, unpublished data), and (d) total nitrogen export (Helmers *et al.*, unpublished data). Measurements were made when soil was not frozen. PFS, prairie filter strips.

mapping techniques[71]; nest searching and monitoring were added in 2010.

Results of the experiment indicated that prairie buffers had disproportionately large effects on water movement, soil loss and nutrient loss, relative to the amount of annual cropland converted to perennial cover. During 2008, 2009 and 2010, greater water runoff occurred in the 100% crop watersheds than in the mixed perennial-annual watersheds (Fig 1a). Concomitantly, soil sediment loss from 100% crop watersheds was at least 10 times greater than from watersheds with 10 or 20% perennial cover (Fig. 1b). Losses of phosphorus (Fig. 1c) and nitrogen (Fig. 1d) in surface runoff were also markedly higher in the 100% crop watersheds. It is notable that no-tillage production techniques, generally viewed as important for soil conservation, failed to prevent large amounts of soil erosion in the 100% crop watersheds, especially in 2008, a year of intense rainfall during late May and early June.

Although differences in plant species richness and cover were not observed among the three treatments that contained prairie vegetation, plant species richness and cover by native perennial species increased with time in the prairie strips. Dominant native perennial species included Canada goldenrod (*Solidago canadensis*), pinnate prairie coneflower (*Ratibida pinnata*), wild bergamot (*Monarda fistulosa*), smooth ox-eye (*Heliopsis helianthoides*), Indiangrass (*S. nutans*) and big bluestem (*A. gerardii*). By 2010, prairie strips contained an average of 51 plant species in a total sample area of 6 m^2, of which 35 were native. In contrast, cropped portions of the watersheds (corn in 2010) contained only 15 plant species other than corn, of which 6 were native. Thus, by 2010, conversion of 10–20% of cropland area to reconstructed prairie gave, on average, a 240% increase in overall plant species richness and a 480% increase in native species richness. Transects within cropped areas indicated that the presence of prairie strips did not increase weed infestation in adjacent crops relative to 100% crop watersheds.

A total of 45 bird species were recorded in the watersheds, including several grassland bird species on Iowa's list of species of greatest conservation need[10]: common yellowthroat (*Geothlypis trichas*), dickcissel (*Spiza americana*), grasshopper sparrow (*Ammodramus savannarum*), lark sparrow (*Chondestes grammacus*), field sparrow (*Spizella pusilla*) and eastern meadowlark (*Sturnella magna*). Bird species richness was higher after prairie establishment (2008–2010) than during establishment (2007), and tended to be highest in watersheds containing 20% prairie in strips (Fig. 2). Individual bird

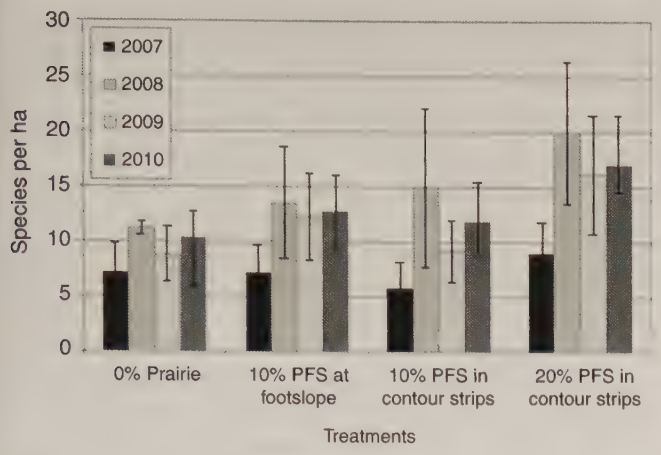

Figure 2. Mean bird species richness in the different experimental treatments, 2007–2010 (Schulte *et al.*, unpublished data). Lines represent one standard error. PFS, prairie filter strips.

species' responses to prairie within the watersheds varied among no response, avoidance, disproportionate use, or dependence on prairie habitat. Many bird species used both prairie and crop vegetation, but with a greater frequency of observations in prairie than expected given the extent of the habitat within a watershed (*G*-values 138–419; $P<0.0001$). In 2010, 17 nests from four bird species were located and monitored, five of which fledged successfully. Although we do not have enough data to robustly assess nest success, success rates for two grassland bird species, dickcissels and vesper sparrows (*Pooecetes gramineus*), are within the range of those observed in other studies[74,75]. Placement of the STRIPs study within the NSNWR may have had some influence on these results, but they are supported by results of a recent study by Walk *et al.*[63], who documented that small patches (1–142 ha) of grassland habitat located within cropland matrices in Illinois could contribute to the nesting success of dickcissels and eastern meadowlarks. In this case, croplands dominated the surrounding landscape (~80% of land cover); furthermore, the investigators did not find a strong influence of patch size on nest success[63].

Taken together, these results provide substantial support for the hypothesis that diversifying small areas of annual cropland through the incorporation of perennial prairie vegetation can have disproportionately large beneficial effects on soil, water and biodiversity conservation. Such effects are especially important because projected climate changes in the US Corn Belt are expected to make farming more challenging. The amount of precipitation from heavy and extreme events is expected to increase in this region during the 21st century[76], with concomitant increases in crop damage[77] and erosion potential[78]. Biodiversity conservation is also expected to become more challenging as rising crop prices encourage producers to convert conservation grasslands to row-crop production[79,81]. In a review of climate change impacts on crop production, Hatfield *et al.*[81] emphasized the need for quantifying how cropping systems can be made more resilient to stress. As shown by results from the STRIPS experiment, strategic targeting to place prairie buffers in landscapes and watersheds dominated by row-crops is likely to be a valuable strategy for pursuing long-term agricultural productivity and environmental quality.

Identifying Trade-offs Among Agroecosystem Performance Indicators: The Comparison of Biofuel Systems (COBS) Experiment

Over the past decade, the USA has had a rapid expansion of interest and investment in transportation fuels derived from plant materials (biofuels), driven in part by concerns over the future availability and costs of fossil energy, and by recognition of the environmental effects of extracting and using fossil energy, including damage from petroleum spills and greenhouse gas emissions. To date, the biofuel boom in the USA has relied almost exclusively on corn grain for ethanol production[82,83]. However, heavy reliance on corn has led to a range of environmental and conservation concerns, including increases in nitrogen and phosphorus contamination of water[84,85], higher rates of soil erosion[86], greater greenhouse gas emissions due to land conversion[87,88], reductions in wildlife habitat and wildlife populations[89,90], and declines in biological control of crop pests[91].

Various forms of perennial vegetation have been proposed as alternatives to corn grain for use as biofuel feedstocks, including trees, certain grasses and mixtures of multiple prairie species[92–94]. These perennials would serve as sources of lignocellulose, rather than starch substrates for fuel production, and might confer a greater degree of environmental protection compared with corn-based systems[95,96]. Previous studies have shown that, in comparison with corn-based systems, perennial plants used as biofuel feedstocks may result in lower nitrate emissions to drainage water[97]; improved soil physical characteristics that can reduce water runoff, soil erosion and phosphorus transport[98]; greater root growth and below-ground carbon storage[98,99]; lower greenhouse gas emissions[100,101]; and better habitat for supporting pollinators, natural enemies of crop insect pests, and other wildlife populations[17,80,87,102].

The wide range of environmental as well as agronomic indicators that can be used to assess biofuel production systems calls attention to the multiple goods (e.g., fuel) and ecosystem services (e.g., water quality protection, carbon sequestration and wildlife conservation) that agricultural systems can provide. The multifunctionality of food and feed production systems in Iowa and Minnesota has previously been investigated by Santelmann *et al.*[103] and Boody *et al.*[104]. It is clear from those studies that no one approach for agricultural

production is likely to score highest for all performance indicators. Similarly, the use of perennial species in lieu of corn for biofuel production may also involve trade-offs among key indicators of system performance.

The COBS experiment was initiated in 2008 at the Iowa State University South Reynoldson Farm in Boone Co., IA, to investigate the productivity characteristics and environmental impacts of corn- and prairie-based biofuel production systems. The 9-ha experimental site was on Mollisols with a relatively flat landscape (largely <1% slope, with some small areas of 2–3% slope), and was used for corn and soybean production prior to the start of the COBS experiment.

The experiment contained five cropping systems: continuous corn grown for grain and stover removal; the same corn-based system, but with rye used as a winter cover crop; a corn–soybean system from which only grain was removed; reconstructed multispecies prairie grown for whole-plant removal, without fertilizer; and the same diverse mixture of prairie species grown for whole-plant removal with fertilizer. The experimental plots were 0.17 ha each and arranged in a randomized complete block design with four replicates. All treatments were managed with no tillage. Planting, fertilization, crop protection and harvest operations were conducted with conventional farm machinery. Both prairie treatments were sown in 2008 with a mixture of 31 species native to Boone Co. or adjacent counties, including cool-season and warm-season grasses, legumes and other forbs. All corn treatments received 84 kg N ha^{-1} at planting, with additional nitrogen applied as a side dressing during early growth, based on treatment-specific soil test results[35]. In 2010, total nitrogen application rates for corn were 105 kg N ha^{-1} in the corn–soybean rotation, 123 kg N ha^{-1} in the continuous corn treatment and 169 kg N ha^{-1} in the continuous corn plus cover crop treatment. The fertilized prairie treatment received 84 kg N ha^{-1} in 2010; no N fertilizer was applied to unfertilized prairie or soybean plots. Weeds were suppressed in corn and soybean plots with post-emergence herbicides; no herbicides were applied to the prairie treatments.

The intent of the COBS experiment was to investigate a wide range of performance indicators. Here, we discuss measurements and calculations made in 2010 related to liquid fuel production, nitrogen loss through leaching, and root growth. Corn and soybean seeds, corn stover and prairie biomass were harvested in late September and October, and potential energy yield per unit of field area (GJ ha^{-1}) was calculated for grain-based ethanol, biodiesel and lignocellulosic ethanol, based on measured yields and conversion data given by Shapouri and Gallagher[105], Conley and Tao[106], Hill *et al.*[107], and Tilman *et al.*[92]. Leaching losses of nitrate–N (kg NO_3–N ha^{-1}) from the different treatments were determined using tile drains installed through the center of each plot at a depth of 1.1 m. Tile lines installed on the borders of each plot were used to prevent subsurface drainage from entering adjacent plots laterally. Flow of water out of central tile lines was metered and flow-proportioned water samples were drawn to monitor nitrate–N concentrations in each plot. After crops were harvested, root biomass (Mg ha^{-1}) was determined using soil cores (80 or 224 cm^2 $plot^{-1}$, depending on soil stratum) taken to a depth of 1.0 m, which were subsequently washed in an elutriator to remove soil particles. Remaining materials were then subjected to flotation and inspection under a dissecting microscope, and recovered roots were dried and weighed.

To facilitate comparisons of the relative performance of the different cropping system treatments, means of each treatment were divided by the highest mean value for each of the three performance indicators: energy production, nitrate–N loss and root mass. Results for 2010, shown in Fig. 3, clearly indicated the existence of trade-offs among different facets of cropping system performance. Energy production (unadjusted for fossil energy used in producing and harvesting crop materials) was greatest from the continuous corn systems, least in the unfertilized prairie system, and intermediate in the fertilized prairie and corn–soybean rotation systems. Nitrate–N loss was greatest from the corn–soybean system, almost nil from both the fertilized and unfertilized prairie systems, and intermediate from the continuous corn systems. The unfertilized prairie system produced more roots than all other systems, including the fertilized prairie system. Root mass of the annual crop systems was ≤14% of the root mass of the unfertilized prairie.

One of the striking features of these results was that the fertilized prairie treatment performed well compared with the conventional corn–soybean system for all three indicators: its estimated fuel yield was similar, it lost less nitrate–N in drainage water and it produced more root mass. Tilman *et al.*[108,109] have shown that above-ground productivity, N retention and root mass can increase with increasing species diversity in prairie communities, due to complementary patterns of resource use among species. High root mass in the prairie treatments of the COBS experiment may have been a consequence of including warm-season (C_4) grasses within the diverse communities, since warm-season grasses can produce more root mass than cool-season (C_3) grasses and legumes[110]. Economic performance indicators obviously need to be considered in any holistic assessment of biofuel feedstock production systems and their potential for adoption. Nonetheless, the potential for prairie-based biofuel systems as ways to address production and environmental challenges merits further investigation.

Using Biodiversity to Link Agricultural Productivity with Environmental Quality at a Landscape Scale

After a half-century of substantial increases in yield and labor efficiency, agriculture in the US Corn Belt has

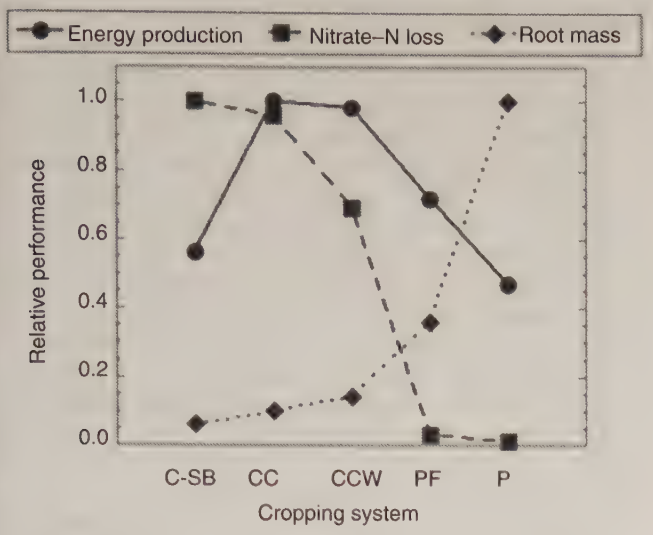

Figure 3. Relative performance of different cropping systems in the COBS experiment in 2010 with respect to liquid fuel production ($GJ\,ha^{-1}$), nitrogen loss in drainage water ($kg\,NO_3\text{–}N\,ha^{-1}$) and root mass ($Mg\,ha^{-1}$). Data for the C-SB treatment represent the average of the corn and soybean phases of the rotation sequence. C-SB, corn–soybean in rotation; CC, continuous corn; CCW, continuous corn with a rye cover crop; PF, fertilized prairie; P, unfertilized prairie. Data used for the calculations are unpublished and are from M. Jarchow, M. Helmers and R. Dietzel.

entered a new and increasingly challenging era. In addition to increasing demands for food, fiber and biofuels, agriculture is now being asked to provide cleaner water, healthier soil, better wildlife habitat, and a range of other environmental goods and services[111].

In this paper, we have reviewed results from three experiments indicating that strategic timing and placement of perennial crop and non-crop species in agroecosystems dominated by annual crops can have large beneficial effects on agroecosystem functioning. In the Marsden Farm cropping systems experiment, lengthening a simple corn–soybean rotation with small grain and forage crops and recycling nutrients and carbon in the form of manure allowed for 76–84% reductions in the use of synthetic N fertilizer, 85–89% reductions in the use of herbicides and 41–56% reductions in the use of fossil energy, while increasing corn yield 3–5%, increasing soybean yield 9–12%, and maintaining equivalent returns to land and management on a whole rotation basis (Table 1). Results of the STRIPs experiment indicate that diversifying a simple corn–soybean system with small strips and patches of reconstructed prairie vegetation reduced soil and nutrient loss from watershed catchments by >90% (Fig. 1), while also increasing the diversity of native plants and birds (Fig. 2). In the COBS experiment, a fertilized prairie community was found to be capable of producing amounts of liquid fuel comparable to amounts produced by a corn–soybean systems, but with six-fold greater additions to the root C pool and with 97% less nitrate–N released to drainage water (Fig. 3). In addition to the perennial forage crops, prairie buffer strips and prairie biofuel feedstocks we have discussed here, other vegetation types and practices exist within the 'perennial portfolio' for restoring agroecosystem health and function, including riparian buffers, agroforestry plantations and constructed wetlands[18,19]. Site-specific use of the full range of such practices would lead to a patchwork mosaic of diverse crops and non-crop vegetation interwoven on landscape and watershed scales[112,113].

In a world in which rising population and greater affluence have created increased demands for commodity crops such as corn and soybean, should arable land be used differently to achieve adequate levels of food, feed, fuel and fiber production? We believe that it should be. Our argument for increasing biodiversity by moving beyond a narrow base of commodity crops has three parts.

The first part relates to the need to re-couple crop and livestock production. Systems in which livestock are concentrated at high densities and isolated from the land base producing the crops that feed them are often characterized by high levels of pollution[114]. In contrast, integrated crop–livestock systems, including the diversification of cereal-based rotations with forage crops, the local application of manure, and pasture-based production, are likely to play an important role in reducing water contamination, soil erosion and fossil fuel dependence in agroecosystems, while maintaining high levels of productivity and profitability[13,37,114–116]. With appropriate forms of crop diversity and integrated management systems, livestock products might be generated in a manner that permits food security to coexist with ecosystem integrity. Results of the Marsden Farm Cropping Systems experiment provide support for the crop portion of this hypothesis, and indicate that diversification and integration can be important ways to reduce reliance on fossil energy and agrichemicals that can become environmental contaminants.

The second part of the argument relates to the need for resilience in the face of climate change. The currently dominant cropping systems in the US Corn Belt were developed in a climate regime different from the one that is beginning to express itself in the region. The types of high intensity rainfall events that are predicted to occur with increasing frequency are exemplified by the storms and flooding that took place in Iowa in May and June of 2008[117]. These events created severe erosion (defined as loss of $\geq 45\,Mg$ of $soil\,ha^{-1}$) on about 930,000 ha, representing 10% of the state's arable cropland[118]. Existing soil conservation practices helped to prevent more damage, but were inadequate for full protection. If agricultural productivity is to be maintained or improved in the future, new conservation practices, such as the small areas of prairie vegetation used in the STRIPs project, need to be developed, refined and implemented. Patches of

perennial vegetation might be used to provide biofuel feedstocks, as in the COBS project, allowing farmers to harvest a crop with use or market value, while still achieving conservation goals[95,119]. A key research question that needs to be addressed is: how small a proportion of arable land might be converted to perennial cover for various conservation benefits, and how does this proportion vary with site-specific conditions and specific conservation goals?

Our final point relates to the cost and value of increased biodiversity. Currently, agroecosystems are valued almost exclusively for their provisioning services, i.e., their ability to generate food and income. Their ability to conserve soil, protect water quality, provide wildlife habitat and recreational opportunities, and offer other supporting and regulating services is largely overlooked in the marketplace and, in most cases, farmers receive little or no financial compensation for providing these services[120]. The development of markets for organic and grass-based crop and livestock products that command price premiums indicates one approach for compensating farmers for ecosystem services that are linked to particular sets of production practices. Nonetheless, most agricultural land in the USA is used for commodity crops, rather than organic and grass-based production. Between 1995 and 2010, US farmers received $167 billion in federal crop subsidies for a narrow group of commodity crops, compared with $35 billion in federal conservation payments[121]. Shifting commodity crop subsidies toward conservation and ecosystem services payments could provide strong financial incentives for farmers to increase crop and non-crop diversity at targeted locations within agricultural landscapes, while maintaining farm income. Moreover, a shift of support from commodity subsidies to payments for ecosystem services could generate substantial benefits for society as a whole, including a more stable food supply, cleaner water and improved outdoor recreational experiences. Moving in this direction requires an explicit recognition of the multifunctional nature of agroecosystems and the need for a balance among different functions. Politically, such a move may be difficult. Ecologically, we feel it is necessary to insure that agroecosystems are both agronomically and environmentally sustainable.

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Diversification and ecosystem services for conservation agriculture: Outcomes from pastures and integrated crop–livestock systems

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Abstract

Conservation agricultural systems rely on three principles to enhance ecosystem services: (1) minimizing soil disturbance, (2) maximizing soil surface cover and (3) stimulating biological activity. In this paper, we explore the concept of diversity and its role in maximizing ecosystem services from managed grasslands and integrated agricultural systems (i.e., integrated crop–livestock–forage systems) at the field and farm level. We also examine trade-offs that may be involved in realizing greater ecosystem services. Previous research on livestock production systems, particularly in pastureland, has shown improvements in herbage productivity and reduced weed invasion with increased forage diversity but little response in terms of animal production. Managing forage diversity in pastureland requires new tools to guide the selection and placement of plant mixtures across a farm according to site suitability and the goals of the producer. Integrated agricultural systems embrace the concept of dynamic cropping systems, which incorporates a long-term strategy of annual crop sequencing that optimizes crop and soil use options to attain production, economic and resource conservation goals by using sound ecological management principles. Integrating dynamic cropping systems with livestock production increases the complexity of management, but also creates synergies among system components that may improve resilience and sustainability while fulfilling multiple ecosystem functions. Diversified conservation agricultural systems can sustain crop and livestock production and provide additional ecosystem services such as soil C storage, efficient nutrient cycling and conservation of biodiversity.

Key words: cropland, crop rotations, crop sequencing, ecosystem function, integrated agricultural systems, soil biology, soil carbon management

Overview

Greater demands for agriculture to provide food, feed, fiber and fuel have placed increased pressure on farmers to intensify their production practices. Increased use of external inputs and converting marginal land for cropping can compromise ecosystem services obtained from agriculture, especially natural resource conservation, soil health and biodiversity. An alternative would be to develop conservation agricultural systems that exploit synergies of crop diversity to improve ecosystem services. Ecosystem services include the 'benefits humans derive, directly or indirectly, from ecosystem functions'[1] and frequently are grouped into four broad categories of provisioning (e.g., food, feed and fiber), supporting (e.g., nutrient and water cycling), regulating (e.g., water purification) and cultural services (e.g., aesthetic experiences)[2].

Agricultural systems became highly specialized during the 20th century[3] with an accompanying decrease in diversity (Fig. 1). Government programs, reliance on petroleum-based products and improved technology are some of the reasons for increased specialization. This specialization decoupled crop and livestock production

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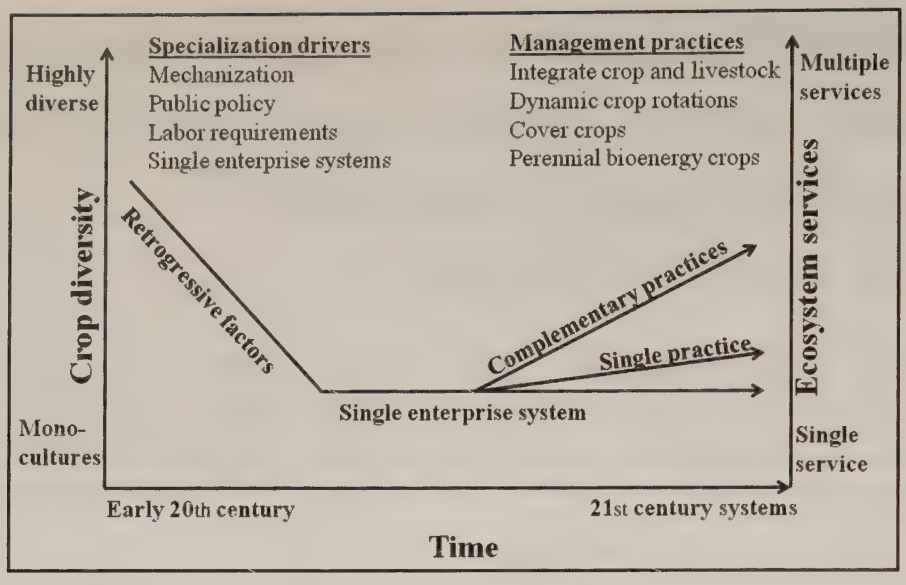

Figure 1. Schematic depicting crop diversity on US farms during the 20th century, the factors associated with the changes and proposed integrated management practices to increase crop diversity and deliver more ecosystem services.

systems[4]. One management alternative to the prevalence of specialized agriculture is integrated agricultural systems at a farm scale. Integrated systems conduct multiple enterprises on the same farm and products from one enterprise are often used by another enterprise[5,6]. Integrated systems also include complementary practices, such as dynamic crop rotations, cover crops and direct utilization of feedstuffs by grazing animals, and can accommodate new enterprises such as perennial bioenergy crops (Fig. 1).

There has been relatively little research on integrated agricultural systems, especially those that include livestock. Integrated crop–livestock research projects are large, not only in land area and livestock requirements but also in financial commitment by research agencies. These projects are often multi-disciplinary in nature and require compromise and cooperation among scientists to ensure that adequate data are collected[7].

As outlined by Franzluebbers[8], conservation agricultural systems rely on three principles to enhance multiple ecosystem services: (1) minimizing soil disturbance, (2) maximizing soil surface cover and (3) stimulating biological activity. Managing for multiple ecosystem services in complex ecosystems (e.g., agroecosystems) requires a high degree of biodiversity in soil organisms, plant communities, and cropping and grazing systems distributed across the landscape[9,10]. We postulate that diverse agricultural systems, especially those that integrate crops and livestock, will conserve natural resources and enhance ecosystem services while maintaining productivity compared with traditional single-enterprise monoculture systems (Fig. 1).

In this paper, we explore the concept of diversity and its role in delivering ecosystem services from managed grasslands and integrated agricultural systems. We begin with managed grasslands and discuss how diversity affects productivity and resistance to weed invasion. Next, we discuss diversified cropping systems including perennial biofuels and then consider the integration of crops and livestock. We then discuss potential outcomes from diversification including the economics of integrated systems along with the trade-offs involved.

Managed Grasslands Diversity and Ecosystem Services

Managed grasslands provide significant provisioning services in the form of food, feed, fiber and fuel. Ecosystem services are enhanced when used as part of conservation approaches such as windbreaks for reducing erosion, vegetative barriers for trapping sediment and reducing water erosion, and buffer strips for protecting riparian zones among other standard practices[11]. Managed grasslands also serve as key reservoirs of biodiversity and can be used for bioenergy production[12].

Plant diversity and grassland productivity

Meta-analyses have pointed to a positive relationship between plant species diversity (most often expressed as species richness) and phytomass production[13] or have shown no clear relationship[14]. A positive diversity–productivity relationship would have obvious practical application in managed grasslands. In that regard, agricultural scientists have begun to explore the relationship in an agronomic context[15].

Greater herbage production from diverse mixtures of forage species compared with monocultures or bicultures (grass–legume) has been demonstrated in multiple

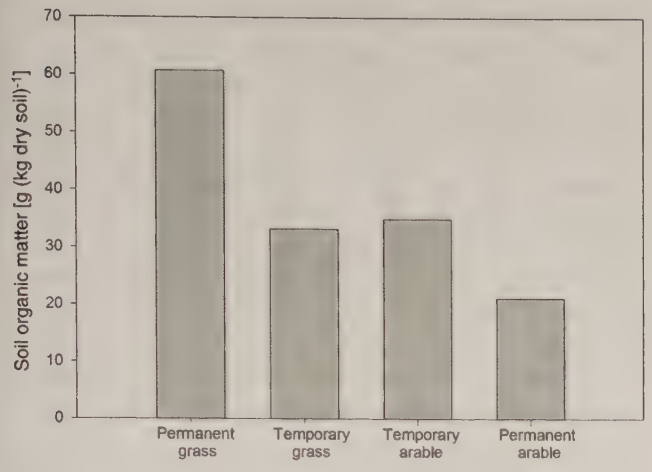

Figure 2. Soil organic matter levels (to a 10-cm depth) in four cropping systems implemented for 36 years in the Netherlands. Permanent grass was perennial ryegrass–white clover mixture. Temporary grassland alternated between 3 years of grass for grazing with 3 years of cropped forage for mechanical harvest. The temporary arable system alternated between 3 years of grass with 3 years of maize. The permanent arable system was in maize. Adapted from van Eekeren et al.[31]

experiments at different scales in the US[16–19] and Europe[20–22]. Economic analyses have also pointed to greater economic returns and reduced production risk compared with grass monocultures or grass–legume bicultures[23,24]. In several instances, the productivity benefit of complex mixtures stemmed from the inclusion of key species (e.g., highly productive or drought tolerant species), which supports the concept that functional traits rather than number of species govern outputs of ecosystem services[25]. The latter point emphasizes the need for more information on functional traits of forages to formulate rational combinations of plants to achieve specific functions in managed grasslands[26].

In pasture-based livestock production systems, greater herbage production and nutritive value are not realized unless the grazing animal efficiently consumes and utilizes the herbage. Grazing trials with livestock have demonstrated variable results for beef cattle gain on forage mixtures versus monocultures[27] and no differences in milk production or herbage intake of dairy cows[28]. Very few grazing trials have been conducted to determine the livestock productivity response to plant diversity in pastures. Greater plant diversity may provide other benefits to grazing livestock including detoxification services[29] and enhanced mineral nutrition[30].

Plant diversity and soil carbon

Managed grasslands generally have greater soil C levels than annual cropland (Fig. 2). For example, managed grasslands in Kansas had 43 Mg ha^{-1} greater soil C in the surface 1 m than annual cropland[32]. There are few, if any, studies on plant diversity effects on C sequestration in forage and pastureland. Research on restored native grasslands at the Cedar Creek, MN site indicated greater soil C storage with greater plant diversity[33]. Grassland restoration practices increased soil C mainly because of the inclusion of the temperate legume red clover (*Trifolium pratense* L.) and not due to increased plant species richness generated by fertilizer reduction or seed addition[33]. Inclusion of red clover may have enhanced nutrient cycling through food webs and improved soil structure leading to increased soil C[34]. Similarly, adding alfalfa (*Medicago sativa* L.) to annual cropping sequences for small grains increased soil C through greater below-ground C input and soil biological activity[35]. Thus, the inclusion of species with key functional traits (e.g., N_2 fixation through a legume) in managed grasslands may be more important than greater richness or diversity. Carbon sequestration in grasslands depends on maintaining C accumulation rates, avoidance of C loss via disturbance and restoring functionality of degraded grassland[36].

Invasion resistance

Invasion resistance of managed grasslands could be improved by formulating unique functional combinations of forage species[37]. Research conducted at several scales in a variety of managed grasslands has indicated that species-rich forage mixtures can reduce weed invasion[17–19,21,38–40]. Combining forage species into functional mixtures has been successful in reducing the establishment of invasive plants in rangelands[41]. Mixed-species pastures may be more resistant to invasion by weeds because of (i) more competition created by greater use of resources or (ii) the inclusion of a few dominant productive species that preclude invasion[42].

Weed abundance was inversely related to the evenness (evenness is a measure of the relative distribution or proportion of plant species in a community) of forage species in clipped plots and on-farm surveys of plant species diversity in pastures[16,43]. In research designed to control for species evenness, however, there was no evidence that evenness of grass–legume mixtures affected weed invasion[44]. Species composition of the mixtures, however, had a strong effect on invasion, indicating that selecting the appropriate species in mixtures to achieve a specific function is more important than evenness of the species in the mixture.

Management tools needed for diverse mixtures

Few, if any studies, have addressed management issues associated with the use of complex mixtures of forages or other crop plants. Tools are needed to aid the selection of specific mixtures for managed grasslands and to guide the spatial distribution or arrangement of species and mixtures across the various site-types on a farm to achieve specific functions. This includes defining the functional traits of plants so that a rational functional diversity

approach can be followed, designing agricultural plant communities to achieve specific functions or combinations of functions rather than simply 'piling on' species to achieve diversity in numbers only[45]. Management of highly diverse grasslands to deliver multiple ecosystem services (e.g., soil C storage, biodiversity conservation and enhanced nutrient cycling) in addition to herbage production will likely require trade-off among services and well-designed economic incentives for farmer adoption[46].

Increasing Cropping System Complexity to Improve the Productivity of Integrated Systems

The foundation of cropping system complexity has its basis in how crops are sequenced over time on the same area of land and where crops are planted across variable landscapes within an individual cropping year. Benefits to agronomic productivity and ecosystem services from increased cropping system complexity are derived from inherent physiological and morphological attributes of crops affecting soil nutrient dynamics, water availability, insect pressure, and incidence of weeds and diseases[47]. By rotating different crop types over time (e.g., grasses, legumes, oilseeds, brassicas, etc.) producers can exploit synergistic relationships among crops to reduce input requirements (e.g., following an N-fixing crop with a high N use crop). Moreover, placement of crops on landscape positions that best suits their growth can improve production efficiencies[47], decrease sediment loss[48] and support biodiversity conservation[49]. Greater attention to synergistic responses within crop rotations, along with increased awareness of crop–landscape complementarity, is crucial to optimize multiple ecosystem functions within integrated systems[50].

Crop selection and sequencing

Crop selection and sequencing can serve as a critical component in conservation of agricultural systems. Among the many options available to select and sequence crops, a fixed-sequence system, whereby crops are sequenced in a consistent, unchanging pattern, is the most simple. Fixed-sequence systems, however, can contribute to the development of pest and disease infestations[51,52], be less responsive to external stressors (e.g., weather)[53] and may limit opportunities to take advantage of market conditions and/or government programs[47]. Accordingly, fixed sequences can suffer from significant drawbacks that limit their sustainability over the long term, particularly in the context of challenges associated with anticipated climate change[54].

To increase cropping system responsiveness to externalities, opportunity/flex crop sequences have been developed to allow for adjustments in cropping intensity and/or diversity using critical information at planting, such as soil water status[53,55]. Even greater flexibility in annual crop sequencing may be realized through the application of a dynamic cropping system concept, where crop sequence decisions are made annually to optimize production, economic and resource conservation goals[56]. Dynamic cropping systems are location specific, with unique crop portfolios (i.e., adapted crop species) within an ecoregion, and require a thorough understanding of short-term (one to three years) crop sequencing effects on relevant agronomic and environmental response variables[57]. Successful application of the dynamic cropping system concept can increase adaptability to high-risk conditions, and may therefore be more economically and environmentally sustainable than other crop selection and sequencing approaches.

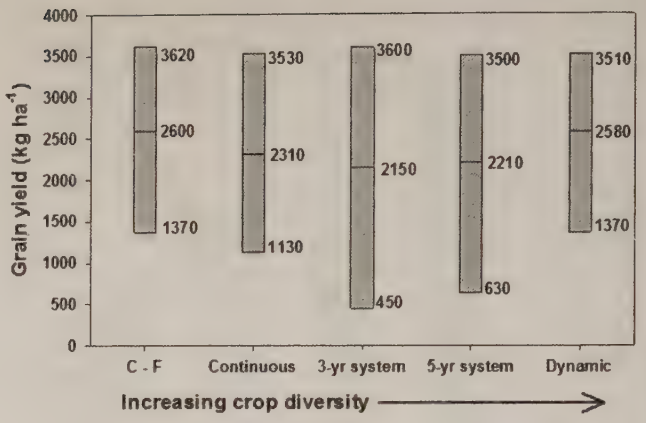

Figure 3. Minimum, maximum and mean spring wheat yield within five cropping systems differing in diversity near Mandan, North Dakota. C-F, Spring wheat–fallow; Continuous, continuous spring wheat; 3-yr system, three-year crop rotation; 5-yr system, five-year crop rotation; Dynamic, dynamic cropping system.

The application of dynamic cropping systems has increased crop yield and decreased yield variability in the northern Great Plains[58]. Strategic inclusion of dry pea (*Pisum sativum* L.) in a dynamic cropping system increased subsequent spring wheat (*Triticum aestivum* L.) grain yield by 30% compared with monoculture spring wheat because of improvements in precipitation-use efficiency[50]. Analysis of long-term data revealed that spring wheat grain yield varied least in a dynamic cropping system compared with other less diverse crop-sequencing approaches[50] (Fig. 3). Moreover, grain yield in the dynamic system was 17–20% greater than 3- and 5-year rotations containing spring wheat (Fig. 3). Such outcomes suggest an inherent resilience of dynamic cropping systems to achieve greater production consistently within a variable environment.

Cover crops

Use of cover crops in annual cropping systems is consistent with the principles of conservation agriculture

from the standpoint of maximizing soil cover, stimulating biological activity and increasing diversity. Although cover crops are often categorized by type (e.g., grasses, legumes and brassicas[59]), their use in addressing specific agronomic and/or natural resource issues underscores the value of functional categorization, such as increasing surface cover, assimilating plant nutrients, improving soil–water relations, and decreasing pests and diseases[60,61]. In this regard, numerous reviews have documented the functional value of cover crops to improve ecosystem services, environmental quality and farm profitability across a wide range of growing conditions[59,62–65]. Among these outcomes, cover crops can reduce soil erosion by protecting soil from raindrop impact, improving soil structure and increasing infiltration[66,67]. Cover crops can enhance soil biological abundance and activity[68], thereby contributing to increased N mineralization rates and N supplying potential of soil[69,70].

Emerging use of cover crop cocktails (i.e., complex mixtures of functionally diverse cover crops planted concurrently) offers the potential to exploit previously untapped spatial and temporal niches in diverse cropping systems to improve ecosystem services. Unfortunately, generalizations regarding their effectiveness are currently limited by a lack of published findings.

Bioenergy cropping systems

Before the 20th century, significant pasture and hayland areas were required for grazing animals to maintain draft power for rural farms and urban transportation[71]. The transition from draft power to the internal combustion engine in US agriculture contributed to the decline in land dedicated to perennial systems, increased reliance on fossil fuels to produce food and led to single enterprise systems (Fig. 1). Concerns about energy security and increased greenhouse gas emissions from fossil fuels have boosted efforts to develop sustainable energy and transportation fuels from biomass. A greater reliance on biofuels may improve or worsen the long-term sustainability of arable cropland, biodiversity and human health, depending on the feedstock source and management practices implemented.

Maize (*Zea mays* L.) grain ethanol is currently the dominant biofuel feedstock in the US and had contributed to annual crop expansion into existing grasslands to meet the demands for food, feed and biofuels[72]. Large-scale conversion of grasslands to maize production can have potentially large effects on biodiversity, water quality, wildlife habitat and other ecosystem services (Fig. 4). Agricultural residues, such as maize stover, are expected to provide substantial amounts of biomass for conversion into biofuels, but excessive maize stover removal can lead to increased soil erosion and decreased soil organic C[73]. Further, grazing of maize stover after harvest is a simple and economical way of integrating crop–livestock systems[78].

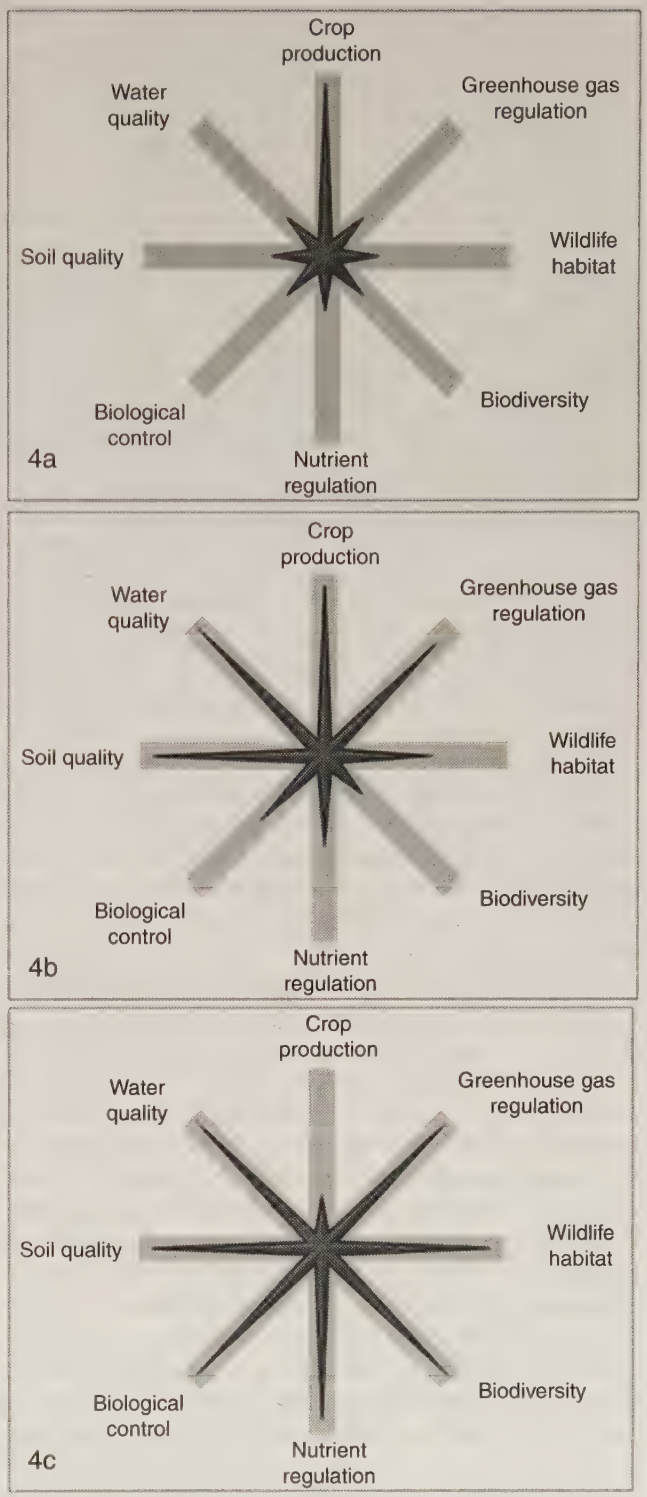

Figure 4. (a) Star diagram showing degree of ecosystem services associated with maize grain ethanol system. The length of each spoke shows the relative magnitude of each ecosystem service provided by the system relative to the highest provision of that service across all systems. (b) Star diagram showing degree of ecosystem services associated with switchgrass bioenergy system. (c) Star diagram showing degree of ecosystem services associated with low-input high-diversity bioenergy system. Source of information based on interpretation of information in selected references[72–77].

If crop residues become a significant source of biomass feedstock, there will be a critical need for timely and accurate data on residue cover to ensure sustainability; however, no government program exists to objectively monitor residue cover across large areas[79–81]. The Conservation Technology Information Center (CTIC) holds the most extensive and up-to-date information on residue cover of the US Midwest, but these data are based on 'windshield observation' estimates[82]. Coupled with a strong scientific foundation and proven algorithms, remotely sensed data offer objective and fast measurement of crop residue cover over large areas. Moreover, accessibility to some satellite images is becoming easier as key government agencies offer these data to the public at no cost[83].

Herbaceous perennial bioenergy crop systems are perceived to be more sustainable and provide more ecosystem services than maize grain ethanol or agricultural residues[74] (Fig. 4). Environmental, societal and economic trade-offs need to be considered when transitioning back to the use of perennial feedstocks to meet rural and urban energy demands. Perennial energy crop systems will probably be planted on marginal cropland or conservation grasslands[84]. Converting annual cropland production to perennial systems will probably reduce soil erosion, increase soil C sequestration, reduce farm-level fossil fuel requirements and increase wildlife habitat[75,76]. However, converting idle, pasture, or conservation land to perennial energy crops may lead to wildlife habitat loss, a decline in plant community diversity and a shift in livestock production from pastures to confinement feeding depending on the region, feedstock used and management practices implemented. Perennial bioenergy cropping systems could also be used as a dual-purpose crop, in which a grazing or forage harvest is implemented in addition to a bioenergy harvest[85]. Rotational systems may be developed in specific climates to incorporate grazing livestock and bioenergy harvests. Integrating a livestock component within herbaceous bioenergy cropping systems further adds management complexity but may offer reduced enterprise risk.

Use of low-input high diversity prairie mixtures (up to 16 species) has been proposed as a sustainable way to produce bioenergy on degraded lands[86] (Fig. 4). Some research demonstrated greater biomass yields with greater numbers of perennial species in mixtures[87]. In other research, however, two-species grass–legume mixtures or N-fertilized switchgrass monocultures produced similar or greater biomass yields than low-input high-diversity mixtures[88,89]. Monocultures of warm-season grasses produced more biomass less expensively than polycultures[90]. Observational research on conservation grasslands in the northeastern USA indicated a negative relationship between plant species diversity on biomass production and biochemical composition conversion[91]. An environmental trade-off exists between maximizing diversity within plant communities grown for bioenergy purposes and having meaningful biomass supplies to meet societal energy demands. However, it should be recognized that native, perennial monocultures or simple polyculture systems provide more ecosystem services such as C sequestration, wildlife habitat, landscape heterogeneity and nutrient regulation than the existing annual cropping systems[92].

Integrated Crop–Livestock Systems

Thus far, we have treated grassland livestock systems and cropping systems separately. World-wide, however, integrated crop–livestock systems (i.e., mixed crop–livestock systems) produce about half of the world's food[93]. There are many ways that livestock and crop production can be integrated. This discussion will focus on integration in the USA where both cattle and annual crops are produced on the same area of land in the same year with cattle grazing crop residue or crops specifically grown for their consumption. This type of crop–livestock integration has both potential benefits and drawbacks. Benefits include adding value to crop residue with little additional input costs, reducing the cost of feeding livestock, reducing water requirements for crop production and reducing the environmental impact of livestock production.

An integrated crop–livestock system significantly lowered the average daily cost of wintering cows on the northern Great Plains compared with feeding them hay[94]. Grazing cattle on cover crops during the winter on cropland used to grow cotton (*Gossypium hirsutum* L.) or peanuts (*Arachis hypogea* L.) during the summer increased farm income in the southeastern USA[95]. Nearly 25% less irrigation water was used in an integrated cotton–forage–beef production system on the Texas High Plains compared with a cotton monoculture. The integrated system also had less soil erosion, required fewer chemical and N fertilizer inputs, and had greater rainfall infiltration than the non-integrated system[96]. Pollution of ground water from N leaching and surface water enrichment with phosphorus is a key environmental concern associated with confining large numbers of cattle in feedlots with high animal density[97]. As a consequence of feedlot-based cattle production, large amounts of urine and feces are concentrated on relatively small areas and atmospheric pollutants (ammonia, nitrous oxide, methane, particulates and odoriferous compounds) are emitted from the animals and (or) the feedlot surface. In contrast, integrating cattle with crop production tends to reduce the density of animals on the land and increases the potential for nutrients in feces and urine to be used for beneficial soil and plant development[98–100].

Soil-related outcomes associated with the integration of livestock and annual cropping systems have been variable, though limited work on the topic has been published. Soil organic C and total N have increased or stabilized with the

inclusion of cattle within an annually cropped system or systems with a perennial grass phase[101–104]. Such increases in soil C and N have been associated with greater aggregate stability, labile organic matter pools and infiltration rates, thereby conferring potential benefits to soil function associated with erosion resistance, nutrient cycling and soil water relations[103–106].

Potential drawbacks of integrating crop and cattle production include soil compaction, which reduces crop production, and interference with new crop growth. Interference can occur if crops or crop residues are windrowed to facilitate grazing but are not adequately grazed during fall or winter because they are covered by excessive snow or ice or if they are not as palatable to cattle as expected. Cattle can cause soil compaction if they are allowed to graze corn residue when the soil is wet, so management guidelines encourage residue grazing only when the soil is dry or frozen unless the soil is tilled before seeding[106]. On the cold northern Great Plains, water infiltration rates were not reduced with 3, 6 or 9 years of an integrated cattle–crop production system with late fall and winter grazing[100]. This region of the US has distinctive freeze/thaw and wet/dry cycles that may limit soil compaction when cattle graze crop fields during the late fall and winter with occasional warm and thawing weather. Another potential drawback of integrated crop–livestock systems may be poor distribution of nutrients from manure and urine, leading to uneven plant growth.

Outcomes from Diversification of Cropping and Integrated Agricultural Systems

Fixing nitrogen to reduce external inputs

Legumes contribute N to cropping systems through symbiotic N fixation. Accordingly, legumes play an important role in diverse cropping systems through their effects on plant available N[69]. Agronomic outcomes from the inclusion of legumes in cropping systems, however, depend on their use either as cover or grain crops, as most N in annual legumes is contained in seed. When harvested for seed, N contributions to the soil from annual legumes are negligible or slightly positive[107,108]. Conversely, when annual legumes are used as green manure, significant N can be made available to soil for subsequent crop uptake[69,109].

Legumes can have additional positive crop sequence effects on subsequent crops. Underlying mechanisms contributing to yield responses are highly complex, and are probably the interaction of legume effects on soil water status, N availability and disruption of pest cycles[52,110,111]. Accordingly, association of N contributions from annual legumes with increased crop yield is not always consistent[51,110]. Conversely, perennial legumes, such as alfalfa, can provide significant short-term (3–5 years) N additions to soil[112] and subsequent yield increases in grain crops following forage stand termination[113,114]. Such yield benefits from perennial legumes, however, are not reproducible across all management systems. In regions where limited precipitation seriously limits crop productivity, perennial legumes can create a 'forage-induced drought', resulting in depressed crop yields following stand termination[114].

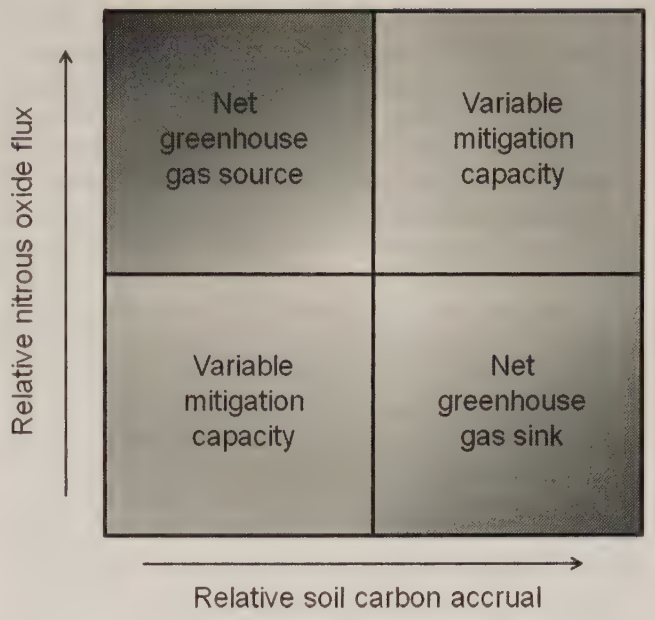

Figure 5. Conceptual diagram documenting potential climate mitigation outcomes associated with soil C accrual and nitrous oxide flux.

Carbon sequestration

Adoption of complex cropping systems can enhance soil C sequestration and improve soil quality. The magnitude and rate of soil C sequestration depends on crop selection and sequence, site history, and edaphic and climatic factors that directly affect biomass productivity and C retention in soil[47,115]. Crop sequences that include perennial phases are most effective at increasing soil C due to below-ground C input from root biomass and rhizodeposition and decreases in soil C loss from erosion[116,117]. Crop sequences with frequent inclusion of easily decomposable, low residue producing crops (e.g., annual legumes) are least likely to increase soil C[118].

It is important to note that while certain cropping systems can lead to soil C accrual, their effects on global warming potential can be variable based on N_2O flux, resulting in either enhanced (e.g., increased soil C and decreased N_2O emission) or negated (e.g., increased soil C and increased N_2O emission) climate mitigation potential (Fig. 5). Such variable responses emphasize the importance of inclusive greenhouse gas evaluations to clearly determine trade-offs associated with management.

Increased soil C in diverse cropping systems has been linked to changes in soil physical, chemical and biological

properties that affect ecosystem functions, such as nutrient cycling, filtering and buffering capacity, and regulation of hydrological attributes[119,120]. Carbon-induced improvements in soil attributes and related functions can positively affect crop yield and environmental quality[121]. Accordingly, management decisions leading to increased soil C in diverse cropping systems contribute to increased agroecosystem resilience under anticipated climate change[122,123].

Enhancing soil biology to improve soil function

Arbuscular mycorrhizal (AM) fungi are critical to the formation of economically and environmentally sustainable grasslands and livestock agricultural systems by closing water and nutrient cycles[124,125]. These fungi are obligate root symbiotes of about 90% of all vascular plants[126] and are ubiquitous in most soils[126–128]. In this relationship, plants allocate photosynthetic C to the fungus to stimulate the acquisition of nutrients from the soil via the fungal hyphae[124,128]. Plants also benefit through greater absorption of water; stimulation of growth hormones; protection against pathogens; and soil structural improvement for better porosity, water infiltration and root penetration[124,129–132].

AM fungal hyphae are also responsible for macroaggregate (>0.25 mm) formation by providing the framework upon which organic matter collects[129]. AM fungi produce glomalin, a glycoprotein found in abundance in both native and agricultural soils[128,130]. Glomalin is located in the fungal hyphal cell wall and may protect it from pathogen attack and nutrient and water loss[131–133]. As the hyphae decompose, glomalin sloughs off and may 'glue' and stabilize soil aggregates[128,130].

In agricultural systems, mycorrhizal fungi are vulnerable to excess fertilization or pesticide use[131,134,135], physical destruction by tillage[136] and absence of host plants, resulting in reductions in C allocation to the rhizosphere[134,137] and weakened relationship with newer crop varieties[138,139]. Excess fertilization may cause plant roots to reject AM colonization, preventing the fungus from obtaining the C it needs to live and grow. Fungicides may reduce mycorrhizal populations directly or indirectly by affecting soil organisms responsible for breaking down organic matter and minerals into the plant nutrients that AM fungi access in exchange for C. Tillage fragments fungal hyphae, making the fungus more susceptible to decomposition and reducing its ability to survive until the next growing season. Use of non-mycorrhizal crops, or crops with a weak relationship with mycorrhizal fungi, limits the amount of C the fungus receives, therefore affecting growth and activity.

Health of the grassland AM fungi relationship depends on plant diversity and community structure, and the level of plant dependence on AM fungi. As grazing intensity increases, mycorrhizal growth and activity decrease through reduction in living roots and activity[140]. Under heavy grazing or no grazing, invasive species tend to dominate, which reduces plant diversity and may reduce mycorrhizal number and diversity[141,142]. Warm-season plants are more dependent on mycorrhizal fungi than cool-season plants[143,144].

Economics

Diversification in agricultural production has economic benefits related to risk management and economy of scope. Economy of scope refers to situations where the cost of producing multiple outputs is lower for a single-integrated firm than for multiple specialized firms[145]. This can occur with complementarities in production, such as in producing crops and livestock, where grain or forages from crop production are used for animal feed, and livestock manure is used for fertilizer, reducing the cost for both. As another example, growing crops in rotation increases yields or reduces inputs relative to growing individual crops continuously in monoculture. Research has shown that the economy of scope in agriculture can be significant. Crops and livestock were produced at a much lower cost for integrated farms in Wisconsin than in specialized farms[146]. The cost of joint crop and livestock production for Missouri farmers was on average 14% less than the cost of specialized crop or livestock production[147]. Integrated farms, however, were less efficient than specialized farms at adjusting the mix of inputs used in production to minimize costs. As a result, total cost efficiency tended to be lower for integrated farms[147]. Diversification can include off-farm income sources as well as different farm enterprises. Off-farm income can be critical for small farms to remain competitive with larger farms[148]. Fernandez-Cornejo et al.[149] showed a 24% cost saving for corn and soybean producers with off-farm income compared with those without.

Both the Wisconsin[146] and Missouri[147] studies showed that the economy of scope tended to decline with the size of the enterprise, so that larger farms would find it beneficial to specialize. This is also probably related to economy of scale. Economy of scale occurs when cost of production decreases with increasing farm size. In developed countries, research indicates that costs tend to decline for small farms and reach a stable lower level for intermediate to large farm sizes[150]. Economy of scale is often associated with specialization, because management of larger farms becomes more complex with more enterprises, and management of more specialized systems is easier and improves production control and efficiency[151,152]. Specialization in larger farms may also contribute to economy of scale by spreading capital costs over more units of production (although this might not be the case if the same equipment/facilities could be spread over multiple enterprises) and the ability to obtain volume discounts on input purchases. However, the trend toward greater specialization with farm size is not universal.

Morrison Paul et al.[153] indicated greater benefits to diversification with larger farm size for US farms. Skolrud et al.[154] showed that the relationship is industry-specific for Washington farms, with trends toward greater diversification with increasing farm size for wheat, beef and apple farms, but greater specialization with increasing farm size for dairies.

Risk management can be an additional driver for diversification. Farmers are generally risk averse, i.e., they are made worse off by being exposed to risk and may be willing to forgo some income for a reduction in risk. Diversification can reduce risk exposure by including income from a range of enterprises that are influenced in different ways by varying weather and market conditions. Incomes from these different enterprises are typically not perfectly correlated, so the overall variation in income is reduced. While risk management concerns may provide an incentive for agricultural diversification, some have noted that diversification to off-farm investments may be a more effective diversification strategy[155]. Other risk management tools including crop insurance and government programs, and marketing tools such as forward contracting and futures and options markets, may have important interactions with diversification. Availability of crop insurance and government programs has been observed to reduce incentives for diversification at the farm level for a South Dakota case farm[156]. Others, however, found a general positive relationship between farm diversification and government payments and crop insurance for US farms[157]. Other factors may be more important drivers of diversification[158]. In addition to risk and economies of scope, other key drivers of diversification include non-uniformity of resource quality (e.g., different soil types might be better suited to different types of production) and resource constraints (e.g., time and equipment constraints), and that the 'main issue raised by variability of price and production is how to respond tactically and dynamically to unfolding opportunities or threats to generate additional income or to avoid losses.'[158] This indicates the need for development of dynamic agricultural systems and tools for managing these systems. This would build on the dynamic cropping systems approach discussed earlier, and expand the approach to other agricultural systems.

Economy of scope may also play an important role in producers' willingness to supply non-marketed ecosystem services. From an economic perspective, the willingness of producers to supply ecosystem services depends on whether provision of these services increases profits or reduces risks at the farm level. Many ecosystem services do not have an effect on profits or risks at the farm level, because many effects are driven by processes at a larger spatial scale than the farm level, occur primarily off-farm and (or) may be public goods where markets do not fully capture the value of these services[159–161]. For example, the spatial pattern of land cover across broad geographic regions can have a large effect on the population and distribution of wildlife species. In the absence of a market for wildlife services, the wildlife produces no income for a farm situated within the landscape. Furthermore, even if a market existed for the wildlife services, the provision of these services would depend on coordinating the pattern of land use among the various farms within the landscape. These services, however, are produced jointly with marketed agricultural products. In cases where the non-marketed ecosystem services are complementary with marketed goods (economy of scope between non-marketed ecosystem services and agricultural products) producers may not need additional incentive to achieve the non-marketed services[162]. This can occur for ecosystem services that provide productivity or input reduction benefits. Examples include practices that enhance pollinator habitat, soil N fixation, or enhance soil quality through increased soil organic carbon (SOC) storage and reduced erosion, and these enhancements can produce benefits beyond the farm level.

Trade-offs

Not all ecosystem services may be realized at once, hence a critical challenge is to identify and accurately assess trade-offs among desired services. In addition, achieving ecosystem service goals may produce economic and social trade-offs. For example, maintaining acceptable wildlife habitat on grassland farms may compromise the quantity and quality of forage needed for livestock production, and this has economic consequences for the livestock producer. In this instance, trade-offs among wildlife habitat, livestock production and economic return at the farm level must be decided and the trade-off assessment encumbered by practical and ethical considerations[163]. The value of an ecosystem service must be assessed within the context of the service, and includes both social and economic values. A grassland farmer may value grassland for its provisioning services (quantity and quality of forage), whereas a grassland conservationist may value attributes such as appropriate plant diversity and vegetation structure that support bird populations. Identifying trade-offs among the various dimensions of sustainability for alternative production systems provide a transparent mechanism for decision makers to determine an appropriate balance among competing goals[164]. Trade-offs can be strongly influenced by spatial variation in resources[165,166], as well as dynamics over time.

Increasing bioenergy production from agricultural land may require trade-offs in land use. In the USA, projections of expanded ethanol and biodiesel production suggest large reductions in pastureland[167]. A global analysis of bioenergy production suggested that shifting livestock production from pasture to confinement feeding would reduce land needs for agriculture[168]. If these changes in land use are realized, there may be pressure to intensify inputs and management on remaining hay, forage and pastureland in the future[169]. The expanding need for

biomass production would probably force forage and grazing livestock production on to more marginal lands with potential trade-offs in soil and water quality.

Monitoring changes in land use, especially in grassland, and in cropping diversity presents a significant challenge. The cropland data layer of the National Agricultural Statistics Service is by far the most extensive spatial dataset depicting the major crop-specific land cover on the entire conterminous USA. But even the cropland data layer generalized grassland into broad categories such as pasture/grass and other hays. Broad categorizations of some of these land uses possess inaccuracy in quantifying ecosystem services and mapping essential resources[170]. A more specific baseline data for monitoring grassland and pastureland would pave the way for better resource management, reliable detection and intervention of land use conversion and accurate assessment of ecosystem services.

An alternative for increasing bioenergy production without shifting land use is utilizing crop residues as feedstocks. A field-level case study analysis of the potential for harvesting corn stover for bioenergy production showed soil organic C losses with corn stover harvest using a traditional tillage system. However, by switching from conventional tillage to no-till, soil organic C could be increased while harvesting biomass and increasing farm profits[77].

Similar cases have been identified where economic returns and provision of ecosystem services may be both improved. For example, in irrigated crop production, research showed opportunities to reduce net greenhouse gas emission and increase economic returns by avoiding excess application of N fertilizer and by switching from conventionally tilled continuous corn to a no-till corn–bean rotation[171]. However, achieving further reductions in net greenhouse gas would reduce farm profitability. This is a common observation where market forces lead producers to maximize profitability, and the economic optimum and environmental optimum typically do not coincide. As a result, achieving increases in ecosystem services often comes at a cost to farm profitability.

This trade-off between ecosystem services and economic returns could be reduced if ecosystem services provided an economic benefit to the farmer. This may occur if markets are developed where farmers would be compensated for the value of ecosystem services they provided. Also, as indicated earlier, this may occur when the ecosystem services provide input-reducing or productivity benefits. Diverse integrated agricultural systems, where inputs from one enterprise come from products of another enterprise, and which rely on well-functioning ecosystem services such as disease and pest suppression, and nutrient and water cycling, are likely to realize these input-reducing and productivity benefits. Diverse integrated systems can reduce the trade-offs, and in some cases provide win–win opportunities for simultaneously increasing agricultural profitability and ecosystem services.

Conclusions

In this synthesis, we demonstrated that diversifying agricultural systems via increasing plant species in managed grasslands and combining several crop species into logical sequences and dynamic rotations can enhance crop production, the use of natural resources and in some instances improve economic returns. However, much of the research documenting potential benefits of diversification is based on studies focused on a single ecosystem service. There is a great need for research comparing various systems and measuring multiple ecosystem services in multiple environments. This will enable analyses of potential trade-offs among services and will better inform management and policy guidelines. Owing to the increased management intensity required by integrated systems based on diversification, agriculturalists need to be convinced that these systems are economically and environmentally sustainable to ensure adoption. Research is needed that quantifies the economic and environmental performance of integrated systems, including both short-term and long-term effects. Ideally, such research would actively engage farmers in the research process and result in useful management guidelines and tools to successfully promote and implement integrated systems.

Achieving global food security would seem to argue for agricultural systems focused solely on production to meet the need for human food. To sustain, and not just meet, food production needs, however, requires the conservation of natural resources that support agricultural systems. Conservation agricultural systems that rely on diversity in plants, animals and soil microorganisms at multiple scales can sustain production and provide additional ecosystem services such as soil C storage, efficient nutrient cycling and conservation of biodiversity.

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Organic grain cropping systems to enhance ecosystem services

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Abstract

Organic grain cropping systems can enhance a number of ecosystem services compared with conventional tilled (CT) systems. Recent results from a limited number of long-term agricultural research (LTAR) studies suggest that organic grain cropping systems can also increase several ecosystem services relative to conventional no-till (NT) cropping systems: soil C sequestration and soil N fertility (N mineralization potential) can be greater while global warming potential (GWP) can be lower in organic systems that use animal manures and cover crops compared with conventional NT systems. However, soil erosion from organic systems and nitrous oxide (N_2O, a greenhouse gas) emissions from manure-based organic systems appear to be greater than from conventional NT systems, though data are limited. Also, crop yields, on average, continue to be lower and labor requirements greater in organic than in both tilled and NT conventional systems. Ecosystem services provided by organic systems may be improved by expanding crop rotations to include greater crop phenological diversity, improving nutrient management, and reducing tillage intensity and frequency. More diverse crop rotations, especially those that include perennial forages, can reduce weed pressure, economic risk, soil erosion, N_2O emissions, animal manure inputs, and soil P loading, while increasing grain yield and soil fertility. Side-dressing animal manures in organic systems may increase corn nitrogen use efficiency and also minimize animal manure inputs. Management practices that reduce tillage frequency and intensity in organic systems are being developed to reduce soil erosion and labor and energy needs. On-going research promises to further augment ecosystem services provided by organic grain cropping systems.

Key words: organic farming, ecosystem services, grain cropping systems, nutrient management, reduced tillage, phenological diversity of crop rotations

Introduction

The primary goal of agriculture is the provisioning of food, feed and fiber. In light of the enormous impact that agriculture and other human activities have on the world's ecosystems and the services they provide, there is now broad interest in demanding that agriculture augment other ecosystem services, such as regulating water quality, climate and pest populations; supporting soil retention and nutrient cycling; and favoring healthy livelihoods and aesthetic experiences[1–3].

Organic farming has been proposed as a means to augment ecosystem services provided by agriculture[4–6]. It may seem intuitive that organic farming should augment ecosystem services compared with conventional systems since organic farming places greater emphasis on managing ecological processes[7,8]. In fact, consumer perceptions that organic farming provides more ecosystem services than conventional agriculture is an important reason that sales of organic food products continue to grow even in the current economic downturn[9]. In the US, for example, the organic food sector grew by 9.4% in 2011 and accounted for about 4% of overall consumer food purchases[10].

The goal of organic farming is to maintain productivity while eliminating those inputs that are, or are perceived to

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Table 1. Characteristics of LTAR projects in the US that include organic (Org) and conventional NT systems (only the treatments with the most similar crop rotations are included for organic treatments; CT systems are listed when present).

Location	Project name	Crop rotations[1]	Animal manure used in organic	Year initiated[2]	Soil types	Representative references
Hickory Corners, Michigan	Kellogg Biological Station Long-Term Ecological Research (KBS LTER)	CT, NT: C-S (1988–1992); C-S-W (1993-present) Org: C-S-W/RC	No	1988	Sandy loam Alfisols	16,17
Arlington and Elkhorn, Wisconsin	Wisconsin Integrated Cropping Systems Trial (WICST)	NT: C-S (CS2) Org: C-S-W/RC (CS3) Org: C-O/P/A-A (CS5)	Yes	1989	Silt loam (Arlington) and silty clay loam (Elkhorn) Mollisols	18,19
Beltsville, Maryland	Sustainable Agriculture Demonstration Project (SADP)	NT: C-W/SOrg: C-W/S (1994–1997); C-S-W/cc (1998–2002)	Yes	1994–2002	Sandy loam Ultisols	20,21
Beltsville, Maryland	Farming Systems Project (FSP)	CT, NT: C-W/S (1996 – 2000); C-r-S-W/S (2001- present) Org: C-r-S-W-v (Org3)	Yes	1996	Silt loam Ultisols	22–28
Kutztown, Pennsylvania	Farming Systems Trial (FST)	CT: C-S NT: C-r-S-W-v Org: C-r-O-B-S-W	No	2008[3]	Silt loam Alfisols	NA[3]

[1] A = alfalfa (*Medicago sativa* L.); B = barley (*Hordeum vulgare* L.); C = corn (maize; *Zea mays* L.); cc = crimson clover (*Trifolium incarnatum* L.) cover crop; O = oats (*Avena sativa* L.); P = pea (*Pisum sativum* L.); r = cereal rye (*Secale cereale* L.) cover crop; RC = red clover (*Trifolium pratense* L.); S = soybean (*Glycine max* L.); v = hairy vetch (*Vicia villosa* Roth.) cover crop; W = winter wheat (*Triticum aestivum* L.).

[2] Termination date included where relevant.

[3] NT treatments were added in 2008 to the FST, which was initiated in 1981. No results from this expanded experiment are yet published.

be, harmful to the environment and human health: synthetic fertilizers, pesticides and genetically modified organisms (GMOs). In the absence of these common agronomic tools, organic farming relies heavily on diverse crop rotations that include cover crops, animal manures and by-products, and tillage to provide soil fertility, weed, insect and disease management, and soil erosion control. While these tools are not unique to organic systems, their importance is elevated in organic farming. Some authors have strongly criticized organic farming since some tools proven to increase crop yields and reduce soil erosion in conventional agriculture are prohibited from organic production[11–14]. These authors argue that lower crop yields in organic systems result in greater deforestation and loss of biodiversity when land is converted to agricultural uses to maintain production at a given level. They also argue that the relatively intense use of tillage in organic systems increases soil erosion.

In this paper, we evaluate the impact of organic farming on the provision of a subset of ecosystem services, focusing on organic grain cropping systems in the US, where the authors have the most experience. While other reviews have focused almost exclusively on comparisons between organic and conventional tilled (CT) systems[4–6], we highlight comparisons between organic and conventional no-till (NT) systems, which are arguably more sustainable than tilled conventional systems[15]. We then explore management practices that can increase ecosystem services provided by organic grain cropping systems: expanding crop rotation diversity, improving manure management, and reducing tillage intensity and frequency.

In assessing ecosystem services, we rely strongly on research results from long-term agricultural research sites (LTARs) since an inherent aspect of organic farming is to build soil quality over the long term. In addition, many ecosystem services are best quantified over the long term to account for changes that occur slowly (e.g., C sequestration) and for variables that can have large interannual variability (e.g., crop yields and greenhouse gas emissions). We know of only five LTARs that include a comparison of organic and conventional NT cropping systems, all in the US; three of these also include a CT system (Table 1). Only six US LTARs include comparisons of organic systems with various crop rotations (Table 2).

Ecosystem Services Provided by Organic and Conventional Systems

Soil organic matter (SOM)

One of the fundamental goals of organic farming is to increase or maintain SOM levels to improve system resilience and resistance to perturbation, enhance nutrient cycling, provide healthy and productive crops, and

Table 2. Characteristics of LTAR projects in the US that include organic crop rotations with varying crop phenologies (other treatments within LTARs are not included).

Location	Name	Crop rotations[1]	Additional information	Year initiated[2]	Soil types	Representative references
Kutztown, Pennsylvania	Farming Systems Trial (FST)	3-yr: C-B-S-O/RC (1986–1990); C-r-S-W/v (1991–2002) 5-yr: C-r-S-r-CS-W/RC + A		1981	Silt loam Alfisols	4,29–31
Lamberton, Minnesota	Variable Input Cropping Management Systems 1 and 2 (VICMS1 and VICMS2)	2-yr: C-S 4-yr: O/A-A-C-S	Without (VICMS1) and with (VICMS2) history of fertilizer and pesticide use	1989 (VICMS1) 1989–2002 (VICMS2)	Clay loam Mollisols	32–35
Arlington and Elkhorn, Wisconsin	Wisconsin Integrated Cropping Systems Trial (WICST)	3-yr: C-S-W/RC (CS3) 5-yr: C-O/P/A-A (CS5)		1989	Silt loam (Arlington) and Silty clay loam (Elkhorn) Mollisols	18,19
Beltsville, Maryland	Farming Systems Project (FSP)	2-yr: C-r-S-v (Org2) 3-yr: C-r-S-W-v (Org3) 6-yr: C-r-S-W/A-A-A (Org6)	Org6 was a 4-yr rotation with a RC and orchardgrass forage prior to 2000	1996	Silt loam Ultisols	22,23,25–28
Greenfield, Iowa	Neely-Kinyon Long-Term Agroecology Research Site (Neely-Kinyon LTAR)	3-yr: C-r-S-O/A 4-yr: C-r-S-O/A-A		1998	Silty clay loam Mollisols	36,37
Morris, Minnesota	No name	2-yr: C-S 4-yr: C-S-W/A-A		2002	Loam, clay loam, silty clay loam Mollisols	38,39

[1] A = alfalfa (*Medicago sativa* L.); B = barley (*Hordeum vulgare* L.); C = corn (maize; *Zea mays* L.); CS = corn silage; O = oats (*Avena sativa* L.); P = pea (*Pisum sativum* L.); r = cereal rye (*Secale cereale* L.) cover crop; RC = red clover (*Trifolium pratense* L.); S = soybean (*Glycine max* L.); v = hairy vetch (*Vicia villosa* Roth.) cover crop; W = winter wheat (*Triticum aestivum* L.).
[2] Termination date included where relevant.

control pests[7,8,40]. Many studies confirm that organic grain cropping systems can increase soil organic carbon (SOC) and total N relative to CT systems[32,41–43], regardless of whether the organic systems rely on legume cover crops alone or in combination with animal manures[44,45]. As noted by Leifeld et al.[46], greater SOC in organic compared with CT systems is likely due to greater C inputs rather than organic management *per se*.

As NT agriculture increases in prominence in the US, Brazil, Argentina, Australia and elsewhere[15], it is important to consider NT systems when comparing ecosystem services provided by organic and conventional systems. In the US, for example, about 35% of row-crop hectares were NT planted in 2009, with the median rate of adoption increasing by about 1.5% per year from 2000 to 2007[47]. Marriott and Wander[45] note that mean change in SOC in nine organic cropping systems they studied was $0.35\,t\,C\,ha^{-1}\,yr^{-1}$, which is similar to that documented for NT systems ($0.36–0.43\,t\,C\,ha^{-1}\,yr^{-1}$) in various regions of the US[48–50].

Differences in SOC were variable from LTARs that include both organic and NT systems, depending on the level of organic inputs. At the W.K. Kellogg Biological Station Long-Term Ecological Research (KBS LTER) site in Michigan, SOC in an organic system ($10.2\,Mg\,C\,ha^{-1}$), to which no animal manure was added, was 22% less than in a NT ($12.4\,Mg\,C\,ha^{-1}$) system after 10 years[16]. Although not reported, carbon inputs, based on aboveground net primary productivity data and management information, appear to be similar in the NT and organic systems. Differences in soil C, then, appear to be due to differences in tillage. SOC was measured only to a depth of 7.5 cm in this study since differences between NT and tilled systems are generally seen at these surface depths.

At the Wisconsin Integrated Cropping Systems Trial (WICST) there was greater SOC at 0–5 cm in a NT corn–soybean (CS2; $26.5\,g\,kg^{-1}$) than an organic corn–soybean–wheat/red clover (CS3; $21.9\,g\,kg^{-1}$) rotation but no differences at 5–20 cm after 18 years[18]. Carbon stocks in the WICST (0–20 cm), calculated from reported soil bulk density and C concentrations, were similar in the NT ($56.0\,Mg\,C\,ha^{-1}$) and organic ($57.6\,Mg\,C\,ha^{-1}$) systems. Even though the organic system (CS3) included cover crops and manure, C inputs were about 25% lower in the organic than the conventional NT system (CS2), due to lower crop residue yields in the organic system, suggesting that the form or placement of C inputs may have impacted soil C stocks.

At the Sustainable Agriculture Demonstration Project (SADP), which was recently concluded in Beltsville, Maryland, Teasdale et al.[20] showed that SOC concentration to a depth of 30 cm was greater in a legume (crimson clover) cover crop plus dairy manure-based reduced-tillage organic system than a NT system after 9 years (13.9 and $10.2\,g\,kg^{-1}$, respectively). Soil C concentration on a volume basis was not reported in this

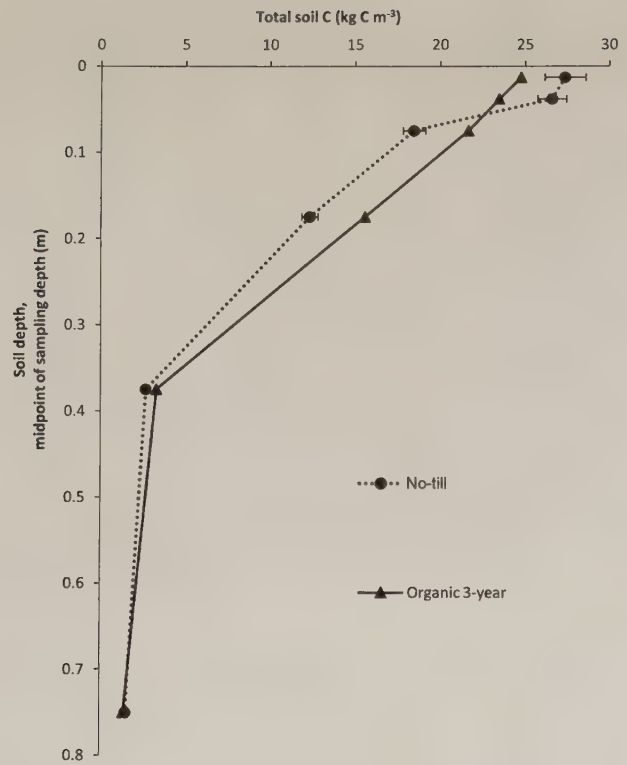

Figure 1. SOC with depth in 3-year NT and organic corn–soybean–wheat/legume crop rotations at the USDA-ARS Beltsville FSP 11 years after the initiation of the experiment.

study. At another LTAR in Beltsville, Maryland, the Farming Systems Project (FSP), SOC to a depth of 1 m in corn–rye–soybean–wheat/legume rotations was 11% greater in a manure-based organic (Org3; 60.8 Mg C ha^{-1}) than in a NT (54.9 Mg C ha^{-1}) system after 11 years[22]. The NT system had not received manure for at least 14 years. Carbon inputs to the soil were greater in the organic than the NT systems in both Maryland studies, largely due to manure and/or compost additions. Results indicate that tilling sufficient organic materials, particularly manure, into soil may be a more effective means of increasing SOC than eliminating tillage.

Distribution of SOC with soil depth also differed substantially between the organic and the NT system at the FSP. While SOC in surface soils (0–5 cm depth) was greater in NT than in the organic system, SOC was substantially greater in Org3 than in NT at 5–10 and 10–25 cm depths (Fig. 1). Burying C inputs (poultry litter, ~4.5 Mg ha^{-1} every 3 years; and cover crop and crop residues) thus provided protection from repeated tillage in Org3. By contrast, SOC in the surface of NT systems is susceptible to loss following tillage[51]. Since the majority of farmers using NT do not use continuous NT[52], results from continuous NT research sites represent an upper limit to C sequestration levels likely achieved on-farm. Results from FSP also suggest that SOC in the organic system at KBS might be greater than indicated by only sampling to 7.5 cm depth. As Doran et al.[53] showed, soil samples should be taken to a minimum depth of 30 cm when comparing SOC in conventional and organic management systems.

SOM and soil fertility

SOM provides a number of supporting and regulating services. Among these benefits, increasing SOM increases soil fertility, reduces global warming potential (GWP) by sequestering atmospheric CO_2 in the soil, and, especially at the soil surface, increases water infiltration and helps stabilize soil to resist erosion.

A number of studies have shown that increasing SOM increases soil N fertility (N mineralization potential) in tilled organic systems[42,43,45,54,55]. The relative impact of organic versus NT management on N fertility in organic systems seems to be related to SOM source and/or input levels. At the KBS LTER, where C inputs appear similar in a NT and an organic system, N mineralization potential was greater in the NT than the organic system after 14 years (http://lter.kbs.msu.edu/datatables/56). As with SOC, N mineralization potential seems driven primarily by differences in tillage when organic matter inputs are similar. At the WICST, although C inputs were greater in a NT than in an organic system, there was no difference in N mineralization potential between the two systems[18]. These results indicate that the quality rather than the quantity of N inputs may have impacted soil N fertility. At the FSP in Maryland, N mineralization potential was greater in the organic than the NT system by 34% after 14 years[23]. Greater soil N fertility resulted in 54% greater corn grain yield in the organic than the NT system in microplots to which no N source was added in year 15. At the SADP, also in Maryland, pre-side-dress soil nitrate, an in-season measure of plant N availability, was 42% greater and corn yield was 18% higher in the organic than the NT system during a uniformity trial following 9 years of experimental treatments[20]. While it is not possible to separate the impact of organic N input level versus source in these studies, it is likely that both factors played a role in augmenting soil N fertility. In any case, results from FSP indicate that N fertility in manure-based organic systems can be augmented relative to NT systems (non-manure based) in the long run even with relatively conservative rates of animal manure application (~4.5 Mg ha^{-1} poultry litter every 3 years).

SOM and GWP

GWP is the balance between the net exchange of the greenhouse gases CO_2, N_2O and CH_4 resulting from on-farm practices and the production and transport of inputs, and is generally driven by changes in SOC and emissions of N_2O in upland cropping systems[56]. GWP is expressed in units of CO_2 equivalents to account for the GWPs of CH_4 and N_2O being 25 and 298 times, respectively, that of CO_2[57]. Rate of change in SOC was the primary factor driving differences in GWP among cropping systems at

both the KBS LTER and FSP sites[16,22]. To our knowledge, these are the only two studies available comparing measured GWP between organic and NT systems; both studies also include CT systems.

At both KBS and FSP, GWP was lower in the organic than the CT system, primarily due to greater SOC in the organic than the CT systems. Other factors contributing to lower GWP in organic than CT included avoiding CO_2 emissions associated with N fertilizer production and transport (indirect energy use) and, at KBS, avoiding CO_2 emissions associated with dissolution of lime applied to soils.

At KBS, GWP was lower in the NT than the organic system, but the opposite was observed at FSP. At KBS, CO_2 emissions from N fertilizer production and transport and from lime dissolution were offset by increases in SOC in NT. GWP was about 2.8-fold lower for the NT than the organic system[16]. By contrast, at FSP, Org3 had greater SOC and lower CO_2 emissions from indirect energy use (energy used to produce and transport agricultural inputs) than NT, which offset approximately two-fold greater N_2O emissions in Org3 than NT. The resulting GWP was negative in Org3 and positive in NT[22]. An important caveat for the FSP study is that poultry litter was assumed to be produced on-farm (transportation distance of 1 km). While this is not an uncommon situation on the Eastern Shore of Maryland, where there is an important broiler chicken industry, manure transport in other locations can be substantial. The CO_2 emissions due to energy use were equal between NT and Org3 if poultry litter was transported 42 km or 114 km for wheat and corn production, respectively[22].

Soil erosion

Soil erosion is sometimes asserted to be lower from organic than from conventional systems due to the erosion-protecting properties of additional SOM in organic systems[4,58,59]. However, as noted by Siegrist et al.[60] reduced soil erodibility in organic compared with conventional systems is not necessarily sufficient to protect against soil erosion during a heavy summer rainstorm.

There are very few direct measurements of soil erosion from organic versus conventional systems. A recent study from England showed lower interrill erosion following simulated rainfall from a silt soil managed organically versus conventionally[61]. Phosphorus content of eroded soil, however, was much greater from the organic than the conventional system. These differences did not reflect soil test P, which was lower in the organically managed soil. One study that quantified the impact of organic compared with conventional farming on soil loss showed that soil depth after 37 years of farming was 21 cm greater on an organic farm than a neighboring conventional farm in the Palouse region of the US[62]. These authors attributed lower soil erosion to greater use of cover crops in the organic system rather than to the soil-erosion controlling properties of additional SOM. Soil erosion is often estimated using mathematical models due to challenges associated with measuring soil erosion directly. When the Water Erosion Prediction Project (WEPP)[63] model was applied to similar 3-year organic (Org3) and CT rotations at the FSP, predicted sediment loss was reduced by 33% in Org3 compared with CT (Fig. 2)[24]. Losses of soil P, N and C in sediment runoff followed a similar pattern. Lower losses in Org3 than CT were due primarily to the presence of a winter legume cover crop following wheat harvest in Org3, a period during which there was no winter cover crop in the CT system. Thus, it seems that reduced soil erosion and nutrient runoff in organic compared with tilled conventional systems is due, at least in part, to greater use of cover crops in organic systems, and the role of greater SOM and associated soil properties is not clear.

While we are not aware of any direct measurements of soil erosion in organic compared with NT systems—which are known to reduce soil erosion substantially compared with CT systems[15]—it is very likely that soil erosion is lower in NT than organic systems due to differences in tillage. When the WEPP model was applied to 3-year rotations at FSP, soil erosion was reduced 80% in NT compared with Org3, with commensurate decreases in losses of soil P, N and C in sediment runoff (Fig. 2)[24]. However, predicted soil erosion from a reduced-tillage organic system was similar to that from NT (3.7 versus 3.5 $Mg\,ha^{-1}$) at the SADP, based on simulations with the EPIC model[21], suggesting that decreased tillage could enhance erosion protection in organic systems.

Crop yield

Crop yield in organic systems has received considerable research attention[13,64,65]. A recent meta-analysis shows that organic grain yields, on average, were lower by 26% than those in conventional systems while organic oilseed yields were similar to those in conventional systems (though variability was high for oilseed data)[66]. As noted by the authors, studies included in this meta-analysis were more rigorously selected than previous studies of this nature.

Lower yields in organic than conventional grain crops are usually related to challenges associated with timely weed control and/or nutrient supply[17,19,25,26,32,33,67]. One study examined this annual variability in detail and found that corn and soybean yields in organic systems were 74% of those in conventional systems in about 1 of 3 years (over 21 site years), largely because of poor weed control in years with wet soils in the spring. By contrast, during the other 2 of 3 years weed control was effective and corn and soybean yields were essentially the same in organic and conventional systems[19]. These results and those of Seufert et al.[66]—showing that organic yields approach those of conventional systems when best management practices are used—indicate that despite a dearth of research on organic systems, organic management has the potential

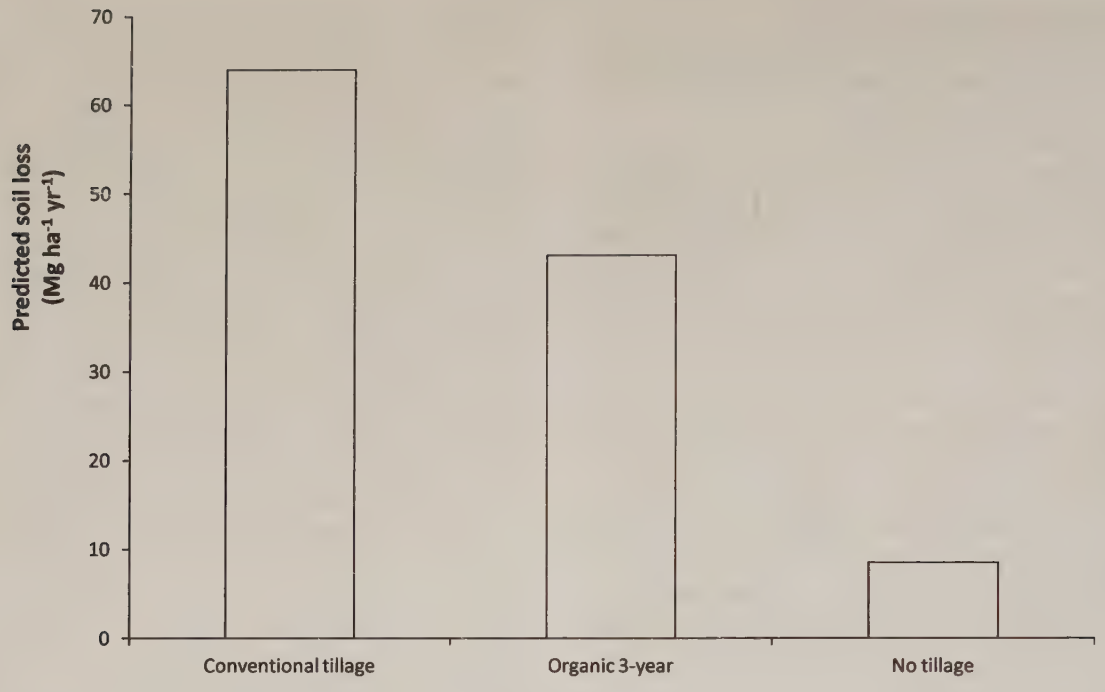

Figure 2. Sediment loss from CT, organic and NT cropping systems at the USDA-ARS FSP, Beltsville, Maryland, USA predicted using the WEPP model. All systems are corn–soybean–wheat/legume rotations; simulations are for a Mattapeake soil on 5% slope, 60 m slope length and 100-year timeline[63].

to produce crop yields similar to those in conventional systems. Additional research and development, including crop breeding, improved agronomic practices and improved engineering of weed control implements, will be needed to improve the consistency of organic system performance.

Some studies have reported greater grain yields in organic than tilled conventional systems during drought years and have attributed these differences to increased soil water-holding capacity in organic systems[4,64], another benefit of elevated SOM. Other studies, which tend to be from the southeast US, show that improved SOM and quality in organic systems does not necessarily result in greater yield in organic systems during dry years[25] (Chris Reberg-Horton, personal communication). Soils in this region of the country are highly weathered, are more vulnerable to crop stress during droughty summer months, and have less tolerance to weed competition than, for example, Midwestern Mollisols. Variability of results highlights the site-specific nature of cropping systems performance and the need for site-specific research to improve organic systems.

Labor requirements and economic performance

Due largely to the multiple tractor passes required to control weeds in organic systems, organic farming has greater labor and management requirements than conventional methods[29,68]. Concerns about these labor requirements are frequently noted by organic grain farmers as a constraint to increasing organic grain acreage[69]. Critical shortage of organic feed corn throughout the US has required organic beef, dairy and poultry producers to purchase feed from distant locations. While the number of organic dairy farms in the US increased by 60% in 2008, the number of organic crop acres increased by only 8%[70]. The US organic poultry industry experienced similar growth[71]. In 2008, certified organic grain cropland was less than 1% of the total acreage while proportions dedicated to corn and soybean were only about 0.2%[72].

The number of grain farmers adopting organic methods has been limited by at least four factors: (1) production challenges, including pest and fertility management[25]; (2) perceived risks of returning to tillage-based systems by conventional grain farmers accustomed to the advantages of NT (Aaron Cooper, Farmer, personal communication); (3) high labor and fuel costs that prevent scaling up (Bill Mason and Ed Fry, Farmers, personal communication); and (4) lack of adequate information on sustainable management practices for organic systems among agricultural professionals, including those working for Cooperative Extension and the Natural Resources Conservation Service (NRCS)[73].

Despite greater labor requirements and frequently lower grain yields, economic returns for organic systems in North America are generally greater than for CT and NT systems on a per hectare basis, due to substantial economic premiums received for organic grain crops[27,34–36,38,39,74–77]. However, on a whole-farm basis, returns for organic farms may be lower than for

Table 3. Impact of increasing crop phenological diversity on corn grain yields, weed pressure, economic risk, soil fertility, reliance on animal manure inputs, soil test phosphorus, soil erosion and soil N_2O emissions at the USDA-ARS Beltsville FSP.

System	Crop rotation[1]	Corn grain yield ($Mg\ ha^{-1}$)[25]	Weed cover (%)[26]	Corn yield loss to weeds (%)[26]	Economic risk[2]	Potentially mineralizable nitrogen ($kg\ ha^{-1}$)[23]	Mean poultry litter application rate ($Mg\ ha^{-1}\ yr^{-1}$)	Soil test P after 16 years ($mg\ kg^{-1}$)[3]	Predicted soil loss rate ($Mg\ ha^{-1}\ yr^{-1}$)[4]	N_2O flux ($kg\ N_2O\text{-}N\ ha^{-1}\ yr^{-1}$)[5]
Org2	C-r-S-v	4.73 c	57	35	87	297 ns	2.2	70	13.0	1.6 a
Org3	C-r-S-W-v	5.55 b	29	17	343	323 ns	3.0	70	7.8	1.6 a
Org6	C-r-S-W-F	6.13 a	18	14	648	325 ns	1.5	55	4.9	0.8 b

[1] C = corn (maize; *Zea mays* L.); r = cereal rye (*Secale cereale* L.) cover crop; S = soybean (*Glycine max* L.); v = hairy vetch (*Vicia villosa* Roth.) cover crop; W = winter wheat (*Triticum aestivum* L.); F, perennial forage crop, which, from 1996 to 2000, was red clover (*Trifolium pratense* L.) and orchardgrass (*Dactylis glomerata* L.); since 2000, has been alfalfa (*Medicago sativa* L.).

[2] 75% lower confidence limit of net returns (a smaller lower limit represents a greater risk), as reported[71].

[3] Mehlich 3 extractable (M.A. Cavigelli, unpublished data).

[4] Calculated using the RUSLE2 for a soil with 2–5% slope and an erodibility factor (K) of 0.28. Results were similar with other settings (T. Pilkowski and M.A. Cavigelli, unpublished data).

[5] Means of measurements made in 2008 and 2010 (Org3) or 2010 only (Org2 and Org6), from[80]. Research supported by GRACEnet, http://www.ars.usda.gov/research/programs/programs.htm?np_code=204&docid=17271.

conventional farms due to the smaller size of organic farms[78]. Thus, labor-saving practices should help increase the number of acres planted to organic grains and improve the economic performance of organic farms.

In summary, results from a limited number of studies indicate that organic cropping systems that include legume cover crops and animal manures can result in greater SOC and soil fertility levels and lower GWP than NT systems, as long as manure transport distance is relatively short. In addition, economic returns on a per hectare basis are usually greater for organic than conventional systems. However, soil erosion is likely lower in NT than tilled organic systems and grain yields and whole-farm economic returns tend to be lower while labor requirements tend to be greater on organic than conventional farms.

Improving Ecosystem Services Provided by Organic Grain Cropping Systems

Can we improve ecosystem services provided by organic cropping systems? Recent and on-going research suggest that ecosystem services from organic systems can be augmented by increasing crop phenological diversity—especially by including perennial forages in the rotation, improving manure management, and reducing the frequency and intensity of tillage.

Increasing Crop Phenological Diversity to Augment Ecosystem Services Provided by Organic Systems

Grain yield

Increasing the phenological diversity of crops in a rotation can increase yields of organic grain crops. At the FSP (Table 2) corn grain yield in a 6-year rotation (Org6) that includes summer annual (corn and soybean), winter annual (winter wheat) and herbaceous perennial (alfalfa) cash crops was, on average, 10% greater than in a 3-year rotation that includes only summer and winter annual cash crops (Org3) and 30% greater than in a 2-year rotation that includes only summer annual cash crops (Org2) (Table 3). These differences were the result of both increases in N availability and decreases in weed competition as crop rotation length and complexity increased[25,26,28]. In Org2, opportunities to kill weeds occur at the same time each year since the two cash crops, corn and soybean, are planted at similar times. Thus, summer annual weeds (primarily *Amaranthus* spp., *Chenopodium album, Daturum stramonium, Setaria* spp., and *Abutilon theophrasti*) are favored in this system. When wheat is added to the rotation (Org3), the summer annual weeds either do not germinate or do not reach reproductive maturity as they are cut prior to setting seed when the wheat is harvested, and killed when soil is prepared for

planting cover crops after wheat harvest. In Org6, a perennial forage crop, alfalfa, provides an additional level of phenological complexity that provides further weed control opportunities. Alfalfa is cut three to five times per year, a disturbance regime that tends to favor perennial and annual grasses with a prostrate growth habit rather than annual broadleaf weeds. Tillage prior to corn planting provides control of the grasses favored during the alfalfa phase of the rotation. Corn yield loss to weeds, as measured in adjacent weed-free and weedy plots, was reduced from 35% in Org2 to 14% in Org6 (Table 3), whereas, to provide context, yield loss to weeds in the 3-year conventional NT rotation was 7%[26].

Organic grain yields also increased with increasing phenological diversity of crops in a rotation at the two Variable Input Cropping Management Systems studies (VICMS1 and VICMS2) in Lamberton, Minnesota (Table 2). Corn grain yield was 26% (VICMS1) and 50% (VICMS2) greater in a 4-year oat/alfalfa–alfalfa–corn–soybean than a 2-year corn–soybean crop rotation[32,33]. Weed pressure was lower at the VICMS2 site, which had a history of pesticide applications, than at the VICMS1 site, which did not have a history of pesticide applications. At VICMS1 there was no impact of rotation length on organic soybean yield but at VICMS2 soybean yield was 42% greater in the 4-year than the 2-year rotation. In Germany, wheat yields in organic systems were found to be 31% greater when a perennial forage was part of the rotation than when only cash crops were part of the rotation[79].

In other studies, phenological diversity of crops in an organic rotation did not impact crop yields. At the Farming Systems Trial (FST) in Kutztown, PA, there was no difference in corn or soybean yields in a 3-year corn–small grain–soybean–small grain–legume rotation compared with a 5-year corn–rye–soybean–silage corn–wheat/red clover–alfalfa rotation over a 16 year time period[30]. At the WICST there was no difference in corn yield between a 3-year corn–soybean–wheat/red clover (CS3) and a 5-year corn–soybean–oat/pea–alfalfa–alfalfa rotation (CS5)[19]. Differences between rotations in these two studies, however, are more subtle than in studies comparing rotations with and without perennial forages (e.g., FSP and VICMS). In Greenfield, Iowa and Morris, Minnesota (Table 2), there were also no impacts of crop phenological diversity during the first 3 or 4 years of experiments comparing 2- or 3- and 4-year rotations[36,37,39]. These results may reflect that these studies report data for only the first 3 or 4 years following plot establishment. Weed populations in organic fields that had previously been managed conventionally are usually low but can increase with time in organic management. In the Iowa study, there are also only subtle differences between the 3- and 4-year rotations (Table 2). While rotations that include perennial forages can increase grain yields compared with simple corn–soybean rotations, there is a need to better understand when and how phenological diversity of crops in organic rotations impacts crop yields.

Economic performance

Increasing crop phenological diversity provides some economic benefits. Crops with different phenology tend to respond differently to weather fluctuations such that the economic performance of a more diverse rotation is better buffered against variable weather. At the FSP, net returns for the 6-year rotation that includes a perennial forage, Org6, were substantially greater than the mean for Org2 and Org3 when no organic price premiums were included. Differences were due to lower production costs and greater returns in the longer rotation[27]. When price premiums for corn, soybeans and wheat were included in the analysis, net returns for the three organic systems were similar and substantially higher than without premiums. However, economic risk, measured using the 75% lower confidence limit of net returns, was 7.5 and 3.9 times greater for Org2 and Org3, respectively, than for Org6 (Table 3). At the VICMS project, Mahoney et al.[34] also found lower risk (and greater return) for a 4-year corn–soybean–oats/alfalfa–alfalfa rotation than a 2-year corn–soybean rotation. On the other hand, researchers in Iowa and Morris, Minnesota found no difference in risk between 2- or 3- and 4-year crop rotations, respectively[36,76]. These results, again, might reflect that these studies report on the first three transition years of these experiments.

Nutrient management

Increasing crop phenological diversity can benefit soil nutrient management. At the FSP, N mineralization potential, particulate organic matter N, and SOC were similar among the three organic systems and all were greater than in CT and conventional NT systems[23]. Interestingly, the longest of the three organic rotations, Org6 relies on fewer external inputs of poultry litter than the two shorter rotations (Org2 and Org3; Table 3). During a 6-year time period, typical poultry litter application rates were 13.4, 17.9 and 9.0 $Mg\,ha^{-1}$ in Org2, Org3 and Org6, respectively. Since P removal in harvested crops was greater in Org6 than Org2 and Org3, soil test P was 21% lower in Org6 than in Org2 and Org3 after 16 years (Table 3). Thus, the possibility of over-loading soils with phosphorus, an important concern in many watersheds, especially when animal manures are applied, was reduced considerably with Org6 compared with the shorter rotations.

Soil erosion

Increasing crop phenological diversity also substantially decreased predicted soil erosion among organic systems at the FSP. When the Revised Universal Soil Loss Equation, Version 2 (RUSLE2) was applied to the three FSP organic systems, predicted soil loss by erosion was reduced by

40 and 62% in Org3 and Org6, respectively, compared with Org2 (Table 3). These results are consistent with the general finding that small grain crops and perennial forages can reduce soil erosion compared with row crops such as corn and soybean[80].

Soil N_2O emissions

Preliminary results from the FSP indicate that annual soil N_2O emissions were reduced by about 50% in the 6-year rotation that includes a perennial forage, Org6, compared with the shorter organic rotations, Org2 and Org3 (Table 3). This represents a N_2O mitigation potential equal to the best mitigation options in agriculture in the Eastern and Central US[81]. This reduction was due to a decrease in the proportion of high N demanding crops in the Org6 rotation compared with the two shorter rotations. Although GWP was not calculated for Org6, reducing N_2O emissions using a more diverse crop rotation that includes proportionally fewer high N-demanding crops should help decrease GWP considerably given that N_2O was a dominant source of GWP at both the KBS LTER and at FSP.

While integrating perennial forages into annual grain cropping systems is an effective way to reduce tillage and improve a number of ecosystem services (Table 3), many producers are reluctant or not able to produce perennial forages due to limitations in equipment, expertise and/or markets. In many areas, development of confined animal feeding operations has concentrated animal production such that large areas have historically low animal populations and forage demand is low[82,83]. Some organic farmers, like their conventional neighbors, are therefore limited to producing primarily grain crops. Thus, there is a need to improve provisioning, regulating and supporting ecosystem services provided by organic rotations that do not include perennial forages[83].

The following sections address research designed to improve ecosystem services provided by organic cropping systems, including improving nutrient management and reducing tillage. Since these are relatively new areas of research, there are fewer data available to assess the viability of these approaches. However, expanding organic grain acreage in the US to meet increasing demand for organic meat and milk products will likely require that we improve existing organic production systems to address farmer concerns with labor (largely tillage) requirements and nutrient management concerns[84] of traditional organic grain cropping systems. These studies highlight promising approaches to improve ecosystem services.

Improving Manure Management in Organic Systems

Since organic farmers often rely heavily on animal manure applications to meet crop N needs, it is imperative that they pay particular attention to nutrient management[84]. The ratio of plant-available N:P in manure or compost (approximately 2:1 and 1:2, respectively) is lower than the ratio of N:P in most crops (between 7:1 and 10:1)[85–87]. Repeated N-based application of animal manure and compost, then, generally leads to an accumulation of soil P in excess of crop needs, thereby increasing the risk of P enrichment of runoff[87–89]. In severe cases, P can also leach through the soil profile[89].

One strategy for optimizing N and P balance on organic farms is to maximize legume-N inputs and thereby reduce animal manure N needs. Intensively managed legume green manures can satisfy a significant portion of crop N demand[90]. In addition, side-dressing or top-dressing crops in-season with manure and other by-products may increase synchrony between N availability and crop N demand. Recent research to evaluate this approach has shown some level of success. In Beltsville, Maryland, researchers applied poultry litter, pelletized poultry litter, feather meal, and a pelletized poultry litter–feather meal blend at side-dress to corn plots where a hairy vetch cover crop had been plowed down. In 2009, side-dress application of all supplemental N materials resulted in a 12% increase in corn yield, a 20% increase in N uptake and a 6% increase in harvest index. In 2010, there were no differences in grain yield or N uptake between pre-plant versus side-dress treatments but harvest index was again 6% higher with side-dress treatments (J.T. Spargo et al., unpublished data). Corn grain yields in side-dress treatments were similar to that in a side-dress ammonium nitrate fertilizer control treatment applied at a similar level of available N. Additional research is needed and there are some constraints to applying organic amendments at side-dress. Nonetheless, better manure management is needed on many organic farms[84].

Reducing Tillage in Organic Systems

Conventional NT management has been recognized for its potential to improve soil quality[91], significantly reduce runoff and soil erosion[92], sequester atmospheric CO_2[49], increase N conservation[52], and reduce machinery, labor, and fuel costs compared with tilled systems[93]. The adoption of NT in conventional systems has been facilitated by the development of improved planters and grain drills, introduction of genetically modified crop germplasm, and the use of effective and affordable broad-spectrum herbicides[94].

By contrast, organic grain production in the Eastern US requires 8–12 tractor operations per year to ensure good weed control (i.e., primary and secondary tillage, seedbed preparation, planting, over the row and between row cultivation). Intensive tillage is an effective weed management tactic, but is energy and labor intensive, and can result in reduced soil quality and increased risk of erosion[95]. While continuous NT is generally considered

Table 4. Soybean yields in cover crop-based organic rotational NT systems in the US. Numbers in parentheses are soybean yields in cover crop-based organic rotational NT systems as percent of soybean yields in counties where research was conducted. County averages were determined using the USDA National Agricultural Statistics Service Quick Stats website (http://quickstats.nass.usda.gov/).

Location	2005	2006	2007	2008	2009	2010	Reference
				kg ha^{-1}			
Urbana, IL	2259 (62)	2713 (73)	1652 (46)				114
Ames, IA		1067 (32)	2724 (81)				112
Rock Springs, PA			5142 (114)	2590 (110)			115
Landisville, PA			5368 (101)	4945 (164)			115
Rock Springs, PA				1100 (48)	2400 (86)		68
Beltsville, MD				2300 (92)	2900 (91)		68
Kutztown, PA				3200 (114)	2000 (56)	1700 (63)	68
Kutztown, PA				3200 (114)			116
Rock Springs, PA					2300 (82)		116
Goldsboro, NC				2190 (101)			95
Kinston, NC				2903 (146)	1112 (48)		95
Plymouth, NC				2694 (114)	2388 (90)		95
Arlington, WI					2885 (103)		117

impractical in organic cropping systems due to perennial weed infestations[68,96,97], organic farmers are eager to develop reduced-tillage organic systems that would combine the soil-protecting capacity of conventional NT systems with the soil-building properties of organic farming[98–101].

Research and development of strategies to reduce tillage in corn and soybean on organic farms is expanding rapidly, although the amount of experimental data available is still small and these practices are not yet widely used. Management approaches are linked to regional differences in climate. In northern US, Canada, and Europe, management focuses on reductions in tillage intensity[102–104]. Since growing seasons are short, it is difficult to successfully integrate cover crops. Also, soil disturbance helps to speed spring soil warming, which facilitates crop germination and establishment, and nitrogen mineralization. In these regions, specialized equipment for shallow or zone tillage has been developed, such as a two-layer plow that inverts the surface soil while only loosening soil lower in the profile[103]. These systems permit subsequent cultivation practices that can control inter- and intra-row weeds. An assessment of this type of system found that soil quality was stratified in the reduced tillage organic system such that microbial biomass C, microbial activity, SOC, and soluble soil P and K were greater in the top 10 cm of soil in the reduced tillage system compared with a conventionally tilled organic system[105].

In areas with longer growing seasons and adequate precipitation, the frequency of tillage in organic grain production can be reduced by using a cover crop-based, rotational NT system. The cover crop-based approach is similar to that practiced by conventional NT farmers in that some crops in the rotation are managed without tillage while others are managed using reduced tillage techniques. Cover crop-based organic rotational NT grain production involves direct seeding of large seeded grain crops into a cover crop that has been killed mechanically and flattened into a cover crop residue mat using a roller-crimper. Winter cover crops are an important component of this system because they occupy a niche otherwise available to weeds in the fall and early spring, and the unincorporated cover crop residue remaining on the soil surface smothers weeds[106]. Planting directly into the flattened cover crop residue extends the duration of a living cover crop in the spring, allowing for greater accumulation of biomass, which is crucial for improving weed control and nitrogen contribution from legume cover crops[107]. The value of other ecosystem services provided by cover crops—preventing loss of sediment and nutrients to the surrounding environment through erosion control[108], building SOM and structure[91], increasing soil water infiltration and storage[109], and enhancing habitat for beneficial organisms[110]—can also increase as the duration of living cover is extended[20,111].

The cover crop-based NT approach can also reduce and redistribute labor and energy requirements compared with standard organic grain crop production[68]. Preparing soil in the fall prior to planting cover crops and NT planting cash crops in the spring redistributes labor and increases management flexibility in the spring. In a recent analysis of energy use in a corn–soybean–wheat rotation, cover crop-based rotational NT required 27% less diesel fuel and 31% less labor than traditional organic management[68]. Reduced management, labor and fuel use have attracted farmer interest in reduced-till organic systems. Further development of these systems could help mitigate regional organic grain shortages by increasing adoption of organic grain production.

Crop performance and grower adoption of organic reduced-till has been greater for soybean than corn phases of crop rotations in the Mid-Atlantic, Southeast,

and Midwest regions of the US[68,112,113]. A summary of reduced-till soybean yield potential in these regions is presented in Table 4. On average, yields for organic reduced-till soybean were 89% of county soybean averages for these sites. This system has been more successful for soybean than corn likely because the system relies on a highly persistent cereal rye cover crop to suppress weeds and a legume cash crop that provides its own N requirements. Weeds are suppressed physically by a thick cereal rye mulch and biogeochemically by soil N immobilization due to the high cover crop C:N ratio (C. Reberg-Horton et al., unpublished data). In contrast, the inability to consistently control weeds and insect pests, and provide adequate fertility has greatly reduced the success of reduced-tillage organic corn systems[68].

Developing a successful reduced-tillage organic corn production system will require an integrated, multi-tactic approach to weed and fertility management[68]. This can be done by designing soil fertility management strategies that meet the agronomic needs of the crop while reducing weed competitiveness. While legume cover crops such as hairy vetch can provide a substantial amount of N to a reduced-tillage corn crop, the residue provides inferior season-long weed suppression compared with vetch/cereal mixtures[118] since legume residues generally decompose more rapidly than those of grasses. A grass–legume mixture should increase weed suppression compared with a legume monoculture, but such mixtures usually reduce release of plant available N compared with a legume monoculture[119,120]. Even when grown as a monoculture, legume cover crops often do not supply sufficient plant available N to satisfy all the needs of a subsequent grain crop when grown alone[90,107,121,122]. Thus, additional fertility sources (e.g., animal manure and by-products) must be considered in organic corn production systems. Manure placement becomes critical in such circumstances as surface applied animal manure and by-products are susceptible to ammonia volatilization, thus decreasing the N:P ratio of amendments and further increasing the risk of soil P loading and potential off-site environmental impacts.

Recent advances in manure placement technology—dairy manure injectors and subsurface poultry litter banders—may provide solutions to this fertility challenge in the cover crop-based, organic rotational NT corn system. By delaying N-mineralization with a cover crop mixture and localizing N placement with subsurface banding of poultry litter, both the timing and placement of N source should favor corn competitiveness against weeds, allow for a more persistent weed suppressive mulch, improve manure NUE, and optimize grain yield and quality.

Conclusions

Current organic grain cropping systems can provide more ecosystem services than their conventional counterparts but, as with all agricultural systems, best management practices are needed to ensure they are managed sustainably. While the past 20 years have seen a great increase in the amount of research conducted on organic farming[123], improving these systems remains in the early stages of development. Promising avenues include increasing phenological diversity of crops in a rotation, improving manure management, and developing rotational NT organic systems that integrate improved cover crop and manure management. Additional research is also needed to improve cash and cover crop varieties for greater yield in organic systems[124,125], address disease[126] and other pest issues, and develop weed control technologies that are less constrained by climate and edaphic conditions than are current tillage options.

Contemporary comparisons between organic and conventional systems are limited by the fact that there has been comparatively little research conducted on organic compared with conventional systems. Current research should continue to evaluate the long-term impacts of organic and conventional cropping systems on the provision of ecosystem services, while using results of this research to develop improved organic cropping systems for the future. Improving organic grain cropping systems should result in increased adoption rates.

Organic cropping systems provide a unique research environment by relying more acutely on ecosystem functions than conventional systems. Lessons learned from organic systems and associated research should be applicable to all systems: strategic soil incorporation of organic matter can increase SOM, which increases inherent soil fertility; increasing crop rotation length increases many ecosystem services; and cover crops provide multiple agro-ecosystem services. At the same time, organic systems can benefit from technologies and practices developed for conventional NT systems.

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Stocker performance and production in mixed tall fescue–bermudagrass pastures of the Southern Piedmont USA

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Abstract

Stocker performance and production from mixed cool- and warm-season perennial pastures are important determinants of agricultural sustainability that can be influenced by management. We evaluated the factorial combination of three sources of nutrient application (inorganic only, organic + inorganic combination, and organic only) and two forage utilization regimes [low grazing pressure (LGP) and high grazing pressure (HGP)] on steer stocking density and rate, performance and production during 7 years of pasture management {tall fescue [*Lolium arundinaceum* (Schreb.) Darbysh.] overseeded into existing Coastal bermudagrass [*Cynodon dactylon* (L.) Pers.] sod} on a Typic Kanhapludult in Georgia, USA. Nutrient source had few major impacts on responses, except for lower animal performance with organic fertilization (broiler litter) than with organic + inorganic and inorganic only fertilization, especially with LGP. Seasonal changes in stocking weight and rate occurred, not only as expected due to environmental conditions and dominant forage species present, but that also counteracted expected differences imposed by grazing pressure; signaling negative feedback of HGP on forage productivity. Steer performance was greatest in spring and summer under both grazing pressures, but was significantly reduced with increasing grazing pressure in the autumn and winter due to low forage availability. Across years, steer gain ha^{-1} (863 kg ha^{-1}) was not different between grazing pressures, but gain ha^{-1} declined with time under HGP and was stable with time under LGP. Reducing grazing pressure to a moderate level can lead to equivalent steer production as HGP, and would likely contribute to a more sustainable balance among production, socio-economic and environmental goals. These multi-year results will help cattle producers in warm, moist climates design and implement more sustainable grazing systems.

Key words: botanical composition, broiler litter, cattle production, grazing pressure, organic fertilizer, stocking density

Introduction

Pastures occupy ~50 Mha of land in the eastern USA, and therefore, are a significant land use affecting the livelihoods of 0.85 million farms[1]. A variety of pasture management systems are employed in the region and there is a pressing need to investigate the long-term effects of grazing management on system ecology, productivity and environmental outcomes[2].

Pastures in the southeastern USA are dominated by two primary perennial forages, i.e. warm-season bermudagrass [*Cynodon dactylon* (L.) Pers.] and cool-season tall fescue [*Lolium arundinaceum* (Schreb.) Darbysh.]. Bermudagrass fits well the hot and variably wet and dry conditions in the summer. In general, bermudagrass is more productive than tall fescue, and therefore fills the primary growing season. However, there is still a need for forage production during the long cool-season when bermudagrass is dormant (November through April). Tall fescue is one of the few perennial, cool-season forages that can withstand the hot, drought-prone conditions in summer.

The fungal endophyte [*Neotyphodium coenophialum* (Morgan-Jones and Gams) Glenn, Bacon, and Hanlin] association with tall fescue is a key determinant of tall fescue stand persistence under high grazing pressure (HGP), as well as of animal performance and production. Cattle responses to different tall fescue cultivars and endophyte associations have been studied in a number of environments and grazing management conditions[3–6]. Wild-endophyte-infected tall fescue persists better with grazing than endophyte-free associations[3,7]. Recently, novel-endophyte associations with tall fescue (producing low levels of ergot-alkaloids) have also proven to be more

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persistent than endophyte-free associations[8–10]. Prior to the development of novel-endophyte associations, persistence of tall fescue in intensively grazed pastures was considered viable only with wild-endophyte association.

'Coastal' bermudagrass is a productive and benchmark hybrid that has been studied in a number of environments and grazing management conditions[11–14]. Bermudagrass hybrids are more productive than common bermudagrass[11] and, therefore, are often recommended to be managed as monoculture forage stands to maximize productivity. In reality though, producers throughout the eastern USA often manage mixed stands of forages, either through intention via planting or default based on evolution of pastures through time from various grazing management strategies employed[15]. Mixed cool- and warm-season forages in the same pasture are expected to extend the grazing season on a particular pasture, confer greater resilience to environmental and management stresses, and could lead to greater sustainability of grass-based agriculture[16].

Investigations of cattle responses to grazing of mixed cool- and warm-season perennial pastures have been limited and, therefore, recommendations are often based on results evaluating the two systems separately. For example in Arkansas, milk yield of cows grazing common bermudagrass in the summer (June through October) was 63 ± 41% greater than that of cows grazing wild-endophyte-infected tall fescue in the fall and spring (November through May)[17]. Seasonal rotation of grazing on separate bermudagrass and tall fescue pastures produced 38 ± 22% greater milk yield than grazing solely on tall fescue. Milk fat was greater from cows rotated between bermudagrass and tall fescue on a seasonal basis than on either pasture alone, but milk protein was not affected by any pasture grazing system[17].

Considering the extent of mixed forage stands of tall fescue and bermudagrass throughout the southeastern USA, there is a scientific void in the characterization of cattle responses to grazing in mixed forage stands. Therefore, our broad goal was to develop balanced, mixed forage stands of tall fescue and bermudagrass for greater year-round grazing. Understanding this complex agro-ecological system would help transfer technologically relevant information to small farmers, especially those with abundant poultry litter available as fertilizer. Yet, the dynamics of cattle responses to nutrient source and forage utilization regimes are largely unknown.

There is concern that high broiler litter application on wild-endophyte-infected tall fescue pastures will exacerbate fescue toxicosis, fescue foot and associated negative responses of animals consuming ergot-alkaloid-containing forage[18,19]. We wanted to test this hypothesis by conducting a multiple-year grazing experiment with low and high broiler-litter application rates against an inorganically fertilized control.

We also hypothesized that reducing grazing pressure would increase cattle performance due to greater forage availability, but would reduce cattle gain ha^{-1} due to reduced stocking rate. Towards the end of 5 years of management of bermudagrass pastures, cattle gain ha^{-1} appeared to have been compromised with HGP. We wanted to test whether this trend would continue with expansion of the grazing season following introduction of tall fescue to the forage base.

Our specific objective was to characterize cattle performance and productivity in response to nutrient source (inorganic only, organic + inorganic, and organic only) and forage utilization [low grazing pressure (LGP) and HGP] in a Coastal bermudagrass sod oversown with Georgia-5 wild-endophyte-infected tall fescue during the course of 7 years in the Piedmont of Georgia, USA.

Materials and Methods

A 15-ha upland field (33°22′N, 83°24′W) on a Typic Kanhapludult near Farmington, Georgia, previously in a 'Coastal' bermudagrass pasture experiment for 5 years[20], was converted to mixed tall fescue–bermudagrass forage composition for this experiment beginning in autumn of 1998. Thirty-six experimental units were arranged in a randomized complete block design with 12 treatments in a split-plot arrangement in each of three blocks. Main plots were nutrient source and split-plots were forage utilization regime.

'Georgia-5' tall fescue with wild-type endophyte infection was drilled into 7-year-old stands of 'Coastal' bermudagrass (5 years differentially managed from 1994 to 1998 following 2 years of establishment in 1992) in November 1998 at a rate of ~19 kg pure live seed ha^{-1}. Prior to seeding in 1998, residual forage mass of bermudagrass in LGP paddocks (~3.5 Mg ha^{-1}) was cut, baled and removed. Forage harvest was not deemed necessary, or practically possible, in paddocks that were maintained under HGP (~1.6 Mg ha^{-1}), since light penetration through the canopy was considered adequate and little forage could be harvested. Drilling of tall fescue (~22 kg pure live seed ha^{-1}) had to be repeated in October 1999 in all paddocks, and again in November 2000 in paddocks maintained under HGP due to extended drought conditions in 1999 and 2000 (Table 1), which limited tall fescue stand establishment.

Nutrient source treatments were: (1) inorganic only; (2) organic + inorganic; and (3) organic only. Nutrient application was targeted to supply 270 kg N ha^{-1} annually in three equally split applications as: (1) inorganic fertilizer with 18–9–18 $N–P_2O_5–K_2O$ broadcast in spring (February–April) and with 34–0–0 (NH_4NO_3) broadcast in summer (May–July) and again in autumn (September–November); (2) chicken broiler litter broadcast in spring and 34–0–0 broadcast in summer and again in autumn; and (3) broiler litter broadcast in spring, summer and autumn. Actual yearly N application rates during the 7 years of this experiment were 261 ± 38 kg N ha^{-1}

Table 1. Climatic conditions during the 1999–2005 study period (10 km from study site at USDA station in Watkinsville, Georgia) and in the long term (1945–2003, Athens airport).

Parameter	Month												
	January	February	March	April	May	June	July	August	September	October	November	December	Yearly
Temperature (°C)													
Long-term mean daily low	0.7	1.9	5.4	9.7	14.4	18.6	20.6	20.1	17.0	10.5	5.3	1.6	10.5
Long-term mean daily high	11.4	13.7	17.8	22.9	27.0	30.4	31.8	31.2	28.0	23.0	17.5	12.5	22.3
Precipitation (mm)													
Long-term mean	119	112	133	96	104	105	124	90	92	79	93	99	1246
1999	152	85	61	49	47	205	78	100	104	81	64	58	1084
2000	124	50	81	39	29	81	119	58	118	7	120	103	929
2001	73	82	225	47	114	145	216	52	30	11	14	43	1052
2002	114	64	165	38	84	82	59	19	155	97	140	153	1170
2003	47	125	164	73	216	143	320	61	84	44	88	66	1431
2004	74	133	25	26	89	171	79	97	303	31	230	61	1319
2005	86	156	186	140	46	274	188	114	3	110	67	101	1471

(inorganic only), 289 ± 36 kg N ha^{-1} (organic + inorganic) and 307 ± 52 kg N ha^{-1} (organic only), varying due to application with commercial-sized equipment and variable nutrient and moisture contents of broiler litter[10].

Forage utilization regimes were two target forage mass levels in grazed paddocks: (1) high forage mass at a target of 2 Mg ha^{-1} (i.e., LGP) and (2) low-forage mass at a target of 1 Mg ha^{-1} (i.e., HGP). This study also had two ungrazed forage utilization regimes in small exclosures, but they were not considered here [(a) unharvested forage, except for an occasional woody plant removal, as a Conservation Reserve Program (CRP) simulation and (b) continuous hay removal on a monthly basis]. Grazed paddocks were 0.69 ± 0.03 ha. Spatial design of paddocks minimized runoff contamination and facilitated handling of cattle through a central roadway. Each paddock contained a 3×4 m shade, mineral feeder and water trough placed in a line 15-m long at the highest elevation.

Yearling Angus steers were used as stock (available herd of 115 ± 24 steers each grazing season; initial age of ~7 months; initial body weight of 249 ± 17 kg in 1999, 239 ± 26 kg in 2000, 212 ± 30 kg in 2001, 229 ± 30 kg in 2002, 217 ± 34 kg in 2003, 214 ± 30 kg in 2004 and 218 ± 21 kg in 2005; weaned just prior to stocking in autumn or wintered on pasture, hay and grain to gain ≥ 0.5 kg d^{-1}). Cattle grazed paddocks nearly yearlong beginning in autumn until target forage mass levels could no longer be maintained, except in 1999 and 2000 when cattle were stocked beginning in spring to avoid grazing on young tall fescue stands. Stocking density was based on achieving the target forage mass of each treatment using a put-and-take grazing system[21] with three tester steers permanently assigned to each paddock within a grazing season and grazer steers added or removed at 28-day intervals. Tester steers were randomly selected from three groups of 18; one group closest to the middle, one group immediately heavier, and one group immediately lighter than the mean weight. All body weight determinations were after 16 h without water while on paddock. Grazer steers were assigned in a similar manner from the remaining pool of animals. Grazer steers not allocated to an experimental paddock grazed an adjacent pasture of similar forage composition.

Before stocking in autumn (or spring starting times in 1999 and 2000), all steers received the following anthelmintic treatment: pour-on eprinex (Ivomec, Merial Limited, Iselin, New Jersey, USA) and oral albendazole (Valbazen, Pfizer, New York, USA) according to label recommendation based on steer weight. Cattle remained in dry-lot with feed and water for 3 days before stocking. Anthelmintic treatment was repeated in spring (April–March).

On initial stocking and restocking days, steers were released into the central roadway early in the morning and corralled together to be weighed. Steer body weight was recorded from a digital balance under a

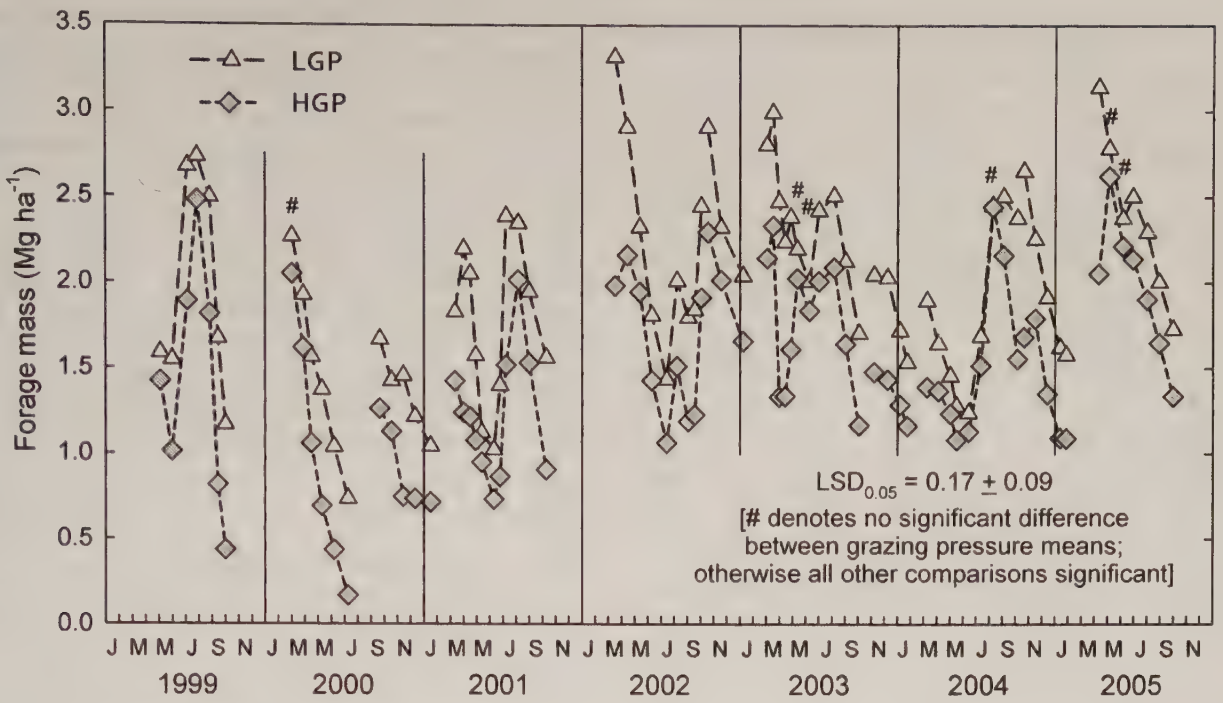

Figure 1. Actual forage mass as affected by grazing pressure level and sampling date within the 7 years of experimentation at Farmington, Georgia. Values are means across nutrient source ($n=3$) levels. LGP, low grazing pressure; HGP. high grazing pressure.

chute. Steers were returned to paddocks mostly within 2 h of corralling.

Steer gain ha^{-1} was calculated as the difference in initial and final body weight of tester steers with proportionality adjustments for total number of steers on a paddock during a grazing period. Steer average daily gain was calculated from the difference in initial and final body weight of tester steers divided by the number of grazing days. Stocking rate was calculated as the mean number of steers on a pasture within the entire season divided by the paddock size (0.65–0.75 ha). Steer stocking weight was calculated from the average body weight of all steers on a paddock (i.e., initial and final body weights).

Response variables were analyzed for variance within each season, year, seasonal means across years, and annual means across years using the general linear model procedure of SAS (SAS Institute Inc., Cary, North Carolina, USA). With the split-plot arrangement of treatments, replication × nutrient source was the error term for the nutrient source effect and replication × nutrient source × forage utilization regime was the error term for forage utilization and nutrient source × forage utilization effects. Grazing season effects were considered a further split-plot in time, evaluated with the experiment-wise error term. All effects were considered significant at $P \leq 0.1$. Although this was a lenient probability level, we did not want to overlook potentially important trends. Actual $Pr > F$ values were also reported for many effects in tables. However, least significant differences were calculated at $P = 0.05$.

Results and Discussion

Environmental conditions

Mean climatic conditions in the Piedmont region of Georgia are relatively mild and supportive of both cool-season and warm-season forages. January is the coldest month (0.7°C minimum and 11.4°C maximum) and July is the warmest month (20.6°C minimum and 31.8°C maximum) (Table 1). Precipitation among months is normally relatively uniformly distributed, 104 ± 16 mm month^{-1}. Precipitation during the 7-year period was 97 ± 16% of normal yearly values. However, precipitation varied greatly from the mean monthly values, resulting in periods of drought and excess precipitation. Compared with 75–125% of the normal monthly precipitation range, there were 45% of months with below-normal precipitation and 26% of months with above-normal precipitation, leaving only 29% of the months with normal precipitation. Extended drought periods (≥3 consecutive months with below-normal precipitation) occurred from March to May 1999 (47% of normal precipitation), February to May 2000 (45% of normal), August to December 2001 (33% of normal), April to August 2002 (54% of normal), and August 2003 to January 2004 (73% of normal).

Residual forage dry mass

Standing forage dry mass throughout the 7-year period varied within a year (Fig. 1), among years, and as

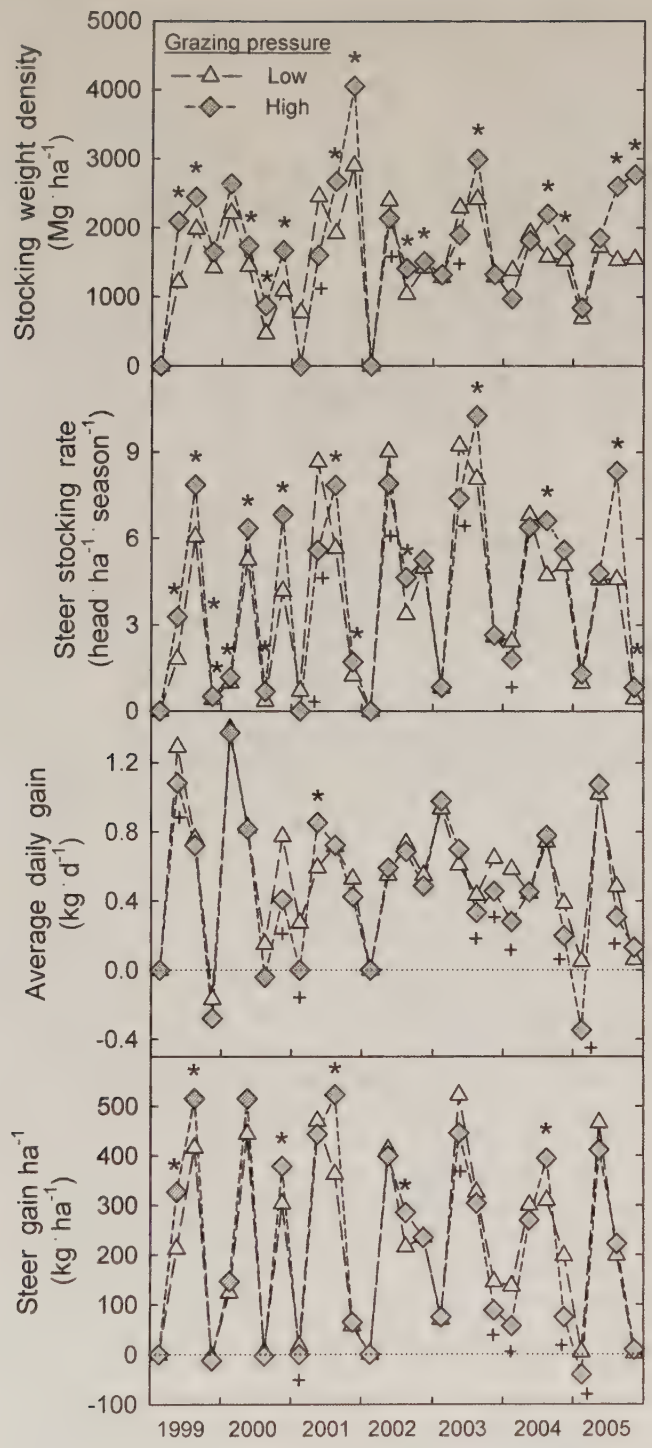

Figure 2. Seasonal changes in steer stocking weight density, stocking rate, average daily gain and gain ha^{-1} as affected by grazing pressure treatment throughout the 7 years of experimentation at Farmington, Georgia. Values are means across nutrient source ($n=3$) levels. *Indicates LGP lower than HGP ($P<0.05$) and +indicates LGP greater than HGP ($P<0.05$).

intended, between grazing pressure treatments. Over the course of the entire 7-year period, there were no differences in standing forage dry mass among nutrient sources (data not shown). Within a total of 77 individual dates of forage dry mass determination, there were only six times when standing forage dry mass was not significantly different between grazing pressure treatments.

Standing forage dry mass varied within and among years due to the relatively long periods (28 days) between planned steer weighing dates, variations in precipitation that altered forage growth rate, and limited number of steers available for stocking. Despite these variations, forage dry mass remained relatively consistently different between grazing pressure treatments throughout (i.e., difference between means was 0.50 ± 0.26 Mg ha^{-1} among the 77 dates). This was a managed difference, and therefore, was intentionally consistent.

Stocking density

Steer stocking density varied within a year, among years, and between grazing pressure treatments (Fig. 2). The effect of nutrient source on stocking density was rarely significant. Stocking density was lower with LGP than with HGP in 14 of 28 seasons, was similar between treatments in 11 seasons, and was greater with LGP than with HGP in three seasons. Across years, stocking density was lower with LGP than with HGP in the autumn when steers were youngest and exposed to pastures for the first time during the year (1.41 versus 1.74 Mg ha^{-1}, respectively; $P<0.001$). Forage dry mass at the beginning of the four autumns in which yearling steers were stocked was 2.32 ± 0.56 Mg ha^{-1} under LGP and 1.68 ± 0.45 Mg ha^{-1} under HGP, values that exceeded our target forage mass levels, and therefore could support differential stocking. In winter and spring, stocking density was reversed in order relative to grazing pressure, in which stocking density was greater with LGP than with HGP (in winter: 1.15 versus 0.94 Mg ha^{-1}, respectively; $P=0.005$; in spring: 1.99 versus 1.85 Mg ha^{-1}, respectively; $P=0.03$). This apparent discrepancy of response relative to imposed treatment regime was a long-term consequence of managing pastures to a target forage mass level rather than on set stocking conditions. In summer, stocking returned to lower density with LGP than with HGP (1.69 versus 2.33 Mg ha^{-1}, respectively; $P<0.001$) due to rapid forage growth. A significant interaction between nutrient source and grazing pressure also occurred in summer, due to greater difference in stocking density between grazing pressure levels with organic fertilization than with inorganic only and organic+inorganic fertilization.

Over the entire 7-year period, stocking density was lower with LGP than with HGP, but the magnitude of difference varied with nutrient source (i.e., 1.72 versus 1.99 Mg ha^{-1} with inorganic fertilization, 1.72 versus 1.91 Mg ha^{-1} with organic+inorganic fertilization, and 1.64 versus 1.99 Mg ha^{-1} with organic fertilization) (Table 2). Steer stocking density varied among years, but this variation was not related to variation in precipitation

Table 2. Steer stocking weight density of each year as affected by nutrient source and forage utilization regimes from 1999 to 2005 at Farmington, Georgia.

Nutrient source	Forage utilization	Year of evaluation							
		1999	2000	2001	2002	2003	2004	2005	7-year mean
		Mg ha^{-1}							
Inorganic	LGP	1.80	1.30	1.96	1.73	2.12	1.69	1.40	1.72
Inorganic	HGP	2.39	1.70	2.01	1.75	2.21	1.81	2.03	1.99
Organic + inorganic	LGP	1.80	1.21	2.14	1.58	2.10	1.66	1.53	1.72
Organic + inorganic	HGP	2.28	1.67	1.97	1.63	2.06	1.73	2.03	1.91
Organic	LGP	1.60	1.19	2.15	1.51	2.03	1.57	1.46	1.64
Organic	HGP	2.20	1.61	2.18	1.68	2.21	1.85	2.18	1.99
LSD ($P=0.05$) among nutrient source × forage utilization means		0.20	0.12	0.18	0.13	0.24	0.12	0.20	0.07
Nutrient source means									
Inorganic		2.09	1.50	1.99	1.74	2.16	1.75	1.72	1.85
Organic + inorganic		2.04	1.44	2.06	1.61	2.08	1.69	1.78	1.81
Organic		1.90	1.40	2.16	1.60	2.12	1.71	1.82	1.82
Forage utilization means									
LGP		2.29	1.66	2.05	1.69	2.16	1.80	2.08	1.96
HGP		1.73	1.23	2.09	1.61	2.08	1.64	1.46	1.69
Source of variation	**df**	*Pr > F*							
Nutrient source (NS)	2	0.42	0.13	0.33	0.13	0.78	0.78	0.81	0.89
Forage utilization (FU)	1	<0.001	<0.001	0.46	0.04	0.22	0.002	<0.001	<0.001
NS × FU	2	0.56	0.69	0.13	0.16	0.34	0.06	0.24	0.04

LPG, low grazing pressure; HGP, high grazing pressure.

among years. The difference in stocking density between grazing pressure treatments was greatest in 1999 and 2005 and there was no difference in stocking density between grazing pressure treatments in 2001 and 2003. Stocking density was a pasture management tool used to manipulate forage to the desired targets we set as part of the treatment structure in this experiment. However, stocking density was also a pasture response reflecting inherent forage productivity dictated by nutrient source effects on nutrient availability and long-term canopy manipulation on pasture growth.

Stocking rate

Steer stocking rate differed from stocking density in this study, because stocking rate was calculated as the number of steers on a pasture during the entire season, not just when pastures were stocked. Stocking rate also varied widely within a year, among years, and between grazing pressure treatments (Fig. 2). Like that of stocking density, the effect of nutrient source on stocking rate was rarely significant. Stocking rate was lower with LGP than with HGP in 15 of 28 seasons, was similar between treatments in only eight seasons, and was greater with LGP than with HGP in five seasons. Stocking rate was lowest in winter, highest in spring and summer, and intermediate in autumn-responses reflecting environmental conditions controlling pasture productivity. Like that of stocking density, stocking rate was greater with LGP than with HGP in the winter (0.8 versus 0.7 steers ha^{-1}, respectively; $P=0.02$) and in the spring (6.5 versus 6.0 steers ha^{-1}, respectively; $P=0.03$), but lower with LGP than with HGP in the summer (4.6 versus 6.6 steers ha^{-1}, respectively; $P<0.001$) and in the autumn (2.7 versus 3.3 steers ha^{-1}, respectively; $P<0.001$).

Seasonal fluctuations in stocking rate were largely a function of available forage for grazing. Under LGP, the greatest stocking rate was in the spring, which was likely due to the large flush of tall fescue growth. Lower stocking rate in summer than in spring with LGP was also likely due to suppression of bermudagrass by tall fescue[22]. Under HGP, the greatest stocking rate was in the summer, which was likely due to greater production of bermudagrass when pastures were maintained with low forage dry mass[22].

Over the entire 7-year period, stocking rate was lower with LGP than with HGP (3.7 versus 4.2 steers ha^{-1}) (Table 3). Steer stocking rate varied among years, and unlike that of stocking density, stocking rate tended to increase with increasing yearly precipitation. This positive association with precipitation was likely related to greater forage production with increasing precipitation. Stocking rate was significantly lower with LGP than with HGP in 1999, 2000, 2004, and 2005, but not different between treatments in 2002 and 2003. In 2001, stocking rate was even greater with LGP than with HGP, but this effect was mainly due to the differential response in organic + inorganic fertilization, whereas no difference occurred

Table 3. Steer stocking rate proportional to the entire year as affected by nutrient source and forage utilization regimes from 1999 to 2005 at Farmington, Georgia.

Nutrient source	Forage utilization	Year of evaluation							
		1999	2000	2001	2002	2003	2004	2005	7-year mean
		Steers ha^{-1}							
Inorganic	LGP	2.2	2.8	3.7	4.6	5.3	4.9	2.5	3.7
Inorganic	HGP	3.0	3.9	3.8	4.7	5.6	5.1	3.7	4.3
Organic + inorganic	LGP	2.1	2.6	4.2	4.2	5.2	4.8	2.8	3.7
Organic + inorganic	HGP	2.9	3.7	3.6	4.2	4.9	5.0	3.7	4.0
Organic	LGP	2.0	2.6	4.2	4.1	5.0	4.6	2.6	3.6
Organic	HGP	2.8	3.7	4.0	4.4	5.3	5.2	4.1	4.2
LSD ($P=0.05$) among nutrient source × forage utilization means		0.2	0.2	0.4	0.3	0.6	0.4	0.5	0.3
Nutrient source means									
Inorganic		2.6	3.3	3.7	4.7	5.5	5.0	3.1	4.0
Organic + inorganic		2.5	3.2	3.9	4.2	5.1	4.9	3.2	3.9
Organic		2.4	3.1	4.1	4.3	5.1	4.9	3.4	3.9
Forage utilization means									
LGP		2.1	2.7	4.1	4.3	5.2	4.8	2.6	3.7
HGP		2.9	3.8	3.8	4.4	5.3	5.1	3.8	4.2
Source of variation	**df**	***Pr > F***							
Nutrient source (NS)	2	0.40	0.09	0.19	0.15	0.47	0.91	0.76	0.76
Forage utilization (FU)	1	<0.001	<0.001	0.03	0.21	0.50	0.01	<0.001	<0.001
NS × FU	2	0.66	0.83	0.06	0.28	0.24	0.16	0.29	0.13

LPG, low grazing pressure; HGP, high grazing pressure.

between grazing pressure treatments with other fertilization regimes. A lone nutrient source effect also occurred in 2000 when stocking rate was greater with inorganic than with organic fertilization. This response was likely due to greater residual forage dry mass that accumulated with inorganic than with organic fertilization in 1999[22], which would have contributed to lower forage availability and stocking rate potential.

Stocking rate was lower during the first phase of this experiment (i.e., 5 years of grazing bermudagrass only from May to September) [Phase 1: 2.2 and 3.3 steers ha^{-1} with LGP and HGP, respectively (yearly values projected from 5.8 and 8.7 steers ha^{-1} during 140 d); Phase 2: 3.7 and 4.2 steers ha^{-1})[20] (Table 3). Extension of the grazing season with the introduction of tall fescue into existing bermudagrass sod allowed more grazing days each year (140 days in Phase 1 and 237 ± 53 days in Phase 2). However, the additional grazing days were associated with lower cool-season production resulting in low-stocking rates in autumn and winter (~ 2 steers ha^{-1}).

Average daily gain

Performance of steers on pasture (i.e., average daily gain; kg d^{-1}) varied widely within a year, among years and between grazing pressure treatments (Fig. 2). Nutrient source also significantly impacted steer performance in several instances. Average daily gain on a seasonal basis was more often similar between grazing pressure treatments (18 of 28 seasons) than different. When differences occurred, 9 of 10 seasons had greater daily gain with LGP than with HGP and only one season had lower daily gain with LGP than with HGP. Variation in average daily gain was particularly high among the 28 continuous seasons, with steer weight loss occurring in three seasons, although all of these observations coincided with very low stocking rate, suggestive of poor grazing conditions with low forage availability. The coefficient of variation among the 28 seasons was lower with LGP (69%) than with HGP (90%), suggesting that weather stresses on steer performance could be somewhat alleviated with LGP. Therefore, adaptability of grazing management to the vagaries of weather conditions could be considered more suitable with LGP by allowing a buffer of residual forage mass to overcome the strong time dependence between forage growth and consumption.

Average daily gain was greater with LGP than with HGP in the autumn (0.55 versus 0.35 kg d^{-1}, respectively; $P=0.001$), in the winter (0.50 versus 0.24 kg d^{-1}, respectively; $P<0.001$), and in the summer (0.63 versus 0.57 kg d^{-1}, respectively; $P=0.006$). In contrast, average daily gain was lower with LGP than with HGP in the spring (0.70 versus 0.76 kg d^{-1}, respectively; $P=0.02$). These seasonal fluctuations in average daily gain suggest that LGP created a forage production—cattle consumption environment that was more stable

Table 4. Steer average daily gain for each year as affected by nutrient source and forage utilization regimes from 1999 to 2005 at Farmington, Georgia.

Nutrient source	Forage utilization	Year of evaluation							
		1999	2000	2001	2002	2003	2004	2005	7-year mean
		kg d^{-1}							
Inorganic	LGP	0.87	0.74	0.69	0.69	0.57	0.56	0.55	0.67
Inorganic	HGP	0.75	0.64	0.65	0.55	0.46	0.45	0.49	0.57
Organic + inorganic	LGP	0.89	0.84	0.57	0.66	0.59	0.55	0.57	0.67
Organic + inorganic	HGP	0.78	0.58	0.75	0.60	0.56	0.45	0.40	0.59
Organic	LGP	0.75	0.70	0.56	0.51	0.54	0.50	0.61	0.60
Organic	HGP	0.72	0.54	0.65	0.64	0.57	0.49	0.48	0.58
LSD ($P=0.05$) among nutrient source × grazing pressure means		0.19	0.12	0.03	0.12	0.06	0.09	0.12	0.03
Nutrient source means									
Inorganic		0.81	0.69	0.67	0.62	0.52	0.50	0.52	0.62
Organic + inorganic		0.83	0.71	0.66	0.63	0.58	0.50	0.49	0.63
Organic		0.74	0.62	0.61	0.58	0.56	0.50	0.54	0.59
Forage utilization means									
LGP		0.84	0.76	0.61	0.62	0.57	0.54	0.58	0.64
HGP		0.75	0.59	0.68	0.60	0.53	0.46	0.46	0.58
Source of variation	**df**	*Pr > F*							
Nutrient source (NS)	2	0.35	0.02	0.18	0.17	0.003	0.90	0.63	0.006
Forage utilization (FU)	1	0.08	<0.001	<0.001	0.48	0.03	0.01	0.006	<0.001
NS × FU	2	0.67	0.15	<0.001	0.02	0.02	0.16	0.40	0.01

LPG, low grazing pressure; HGP, high grazing pressure.

throughout the year than under HGP. The lower steer performance under LGP than under HGP in the spring was likely due to the unexpectedly greater steer stocking rate in the spring to consume the greater tall fescue forage accumulation left after the winter under LGP.

The difference in average daily gain between LGP and HGP treatments in winter was greater with organic + inorganic fertilization (0.54 versus 0.17 kg d^{-1}, respectively) than with inorganic only and organic only (0.47 versus 0.27 kg d^{-1}, respectively), but this effect was simply a matter of magnitude not of order. The difference in average daily gain between LGP and HGP treatments in summer was significant for inorganic only and organic + inorganic fertilization (0.66 versus 0.55 kg d^{-1}, respectively), but not for organic fertilization (0.58 kg d^{-1}). It is unclear why this interaction occurred, since the stocking rate was consistently lower with LGP than with HGP in the summer from all three nutrient source treatments (4.6 versus 6.6 steers ha^{-1}). Average daily gain was also greater with inorganic only and organic + inorganic fertilization (0.75 kg d^{-1}) than with organic only fertilization (0.68 kg d^{-1}) in the spring. Lower daily gain with organic fertilization might have been due to a numerically greater ($P>0.1$) stocking rate than with other fertilization strategies (6.4 versus 6.1 steers ha^{-1}) in the summer.

Over the entire 7-year period, average daily gain was greater with LGP than with HGP (0.64 versus 0.58 kg d^{-1}) (Table 4). Average daily gain tended to decline with time (and precipitation due to covariance). This may have been a consequence of greater time of stocking during winter and autumn in later years of this study than in earlier years. These two periods produced the lowest steer performance, especially under HGP. Consumption of toxic ergot alkaloids in the wild-type-endophyte association of tall fescue was a likely contribution to the depression in cattle performance during the cool seasons[6].

Average daily gain was significantly greater with LGP than with HGP in 1999, 2000, 2003, 2004 and 2005, but not different between treatments in 2002. In 2001, average daily gain was lower with LGP than with HGP in organic + inorganic and organic only fertilization, but greater with LGP than with HGP in inorganic fertilization. A similar disparity in response occurred in 2002 and 2003. It is not clear why broiler litter as part of the fertilization strategy under LGP might have reduced steer performance, but high P input with broiler litter has been hypothesized to exacerbate fescue toxicosis symptoms (J.A. Stuedemann, unpublished data) and this response should be investigated further in other experimental conditions. Nutrient source main effects were also significant in 2000 and 2003, in which organic + inorganic fertilization was greater than other sources. Again, these are curious results that require further mechanistic investigations to fully understand.

Table 5. Steer gain ha^{-1} for each year as affected by nutrient source and forage utilization regimes from 1999 to 2005 at Farmington, Georgia.

Nutrient source	Forage utilization	Year of evaluation							
		1999	2000	2001	2002	2003	2004	2005	7-year-mean
		kg ha^{-1}							
Inorganic	LGP	680	938	970	1000	1081	980	610	894
Inorganic	HGP	859	1080	950	911	808	727	617	850
Organic + inorganic	LGP	671	887	874	872	1107	975	685	868
Organic + inorganic	HGP	852	1035	1056	862	901	795	509	859
Organic	LGP	523	804	878	719	1003	877	718	789
Organic	HGP	779	999	1086	986	1028	865	683	918
LSD ($P=0.05$) among nutrient source × grazing pressure means		188	182	93	192	146	136	197	83
Nutrient source means									
Inorganic		769	1009	960	956	944	854	613	872
Organic + inorganic		762	961	965	867	1004	885	597	863
Organic		651	901	982	852	1016	871	701	853
Forage utilization means									
LGP		625	876	907	864	1064	944	671	850
HGP		830	1038	1030	919	912	796	603	876
Source of variation	**df**	*Pr > F*							
Nutrient source (NS)	2	0.19	0.28	0.88	0.16	0.71	0.73	0.17	0.87
Forage utilization (FU)	1	0.004	0.009	0.001	0.26	0.005	0.004	0.19	0.24
NS × FU	2	0.74	0.86	0.01	0.04	0.03	0.05	0.31	0.02

LPG, low grazing pressure; HGP, high grazing pressure.

Compared with Phase 1 of this experiment, average daily gain declined with introduction of tall fescue into the bermudagrass sod (Phase 1: 0.88 and 0.67 kg d^{-1} with LGP and HGP, respectively, versus Phase 2: 0.64 and 0.58 kg d^{-1}). Considering only the spring and summer grazing seasons in Phase 2, average daily gain was 0.67 kg d^{-1} under both grazing pressures, and therefore, equivalent to that under HGP in Phase 1. Dilution of ergot-alkaloid containing forage with other forages has been a recommended practice for overcoming toxicity issues in tall fescue pastures[23]. Average daily gain when tall fescue was the dominant forage (cool seasons of autumn and winter) was low (0.52 and 0.29 kg d^{-1} with LGP and HGP, respectively), and was likely a consequence of steers consuming ergot-alkaloid containing forage. Average daily gain of heifers grazing tall fescue in a nearby study was 0.54 kg d^{-1} on wild-endophyte association and 0.75 kg d^{-1} on novel-endophyte and endophyte-free associations[24], a clear indication of depression of animal performance with consumption of ergot-alkaloid containing forage. Greatest depression in animal performance occurred in spring and autumn in this neighboring study, followed closely by depression in winter, but there was no effect on animal performance in summer. Interestingly, our results reported here suggested no depressive effect of consuming tall fescue forage in the spring (average daily gain was 0.7 kg d^{-1} in spring and 0.6 kg d^{-1} in summer).

Reduction in steer grazing pressure led to somewhat mixed results over the course of the 7 years of experimentation (Table 4). LGP led to greater steer performance than HGP in 10 of 21 nutrient source–year combinations. As well, no significant differences occurred in eight combinations, but three occurrences led to poorer steer performance. The majority of effects suggest that reducing grazing pressure can improve steer performance.

Steer gain ha^{-1}

A common method of assessing productivity of pastures is with steer gain ha^{-1}, as this response integrates steer stocking rate as a function of forage production and steer performance. Similar to that of stocking rate and average daily gain, steer gain ha^{-1} varied widely within a year, among years and between grazing pressure treatments (Fig. 2). Several significant nutrient source × forage utilization interactions occurred with gain ha^{-1}.

Steer gain ha^{-1} on a seasonal basis was most often similar between grazing pressure treatments (16 of 28 seasons), but there were also six seasons when gain ha^{-1} was lower with LGP than with HGP and six seasons when gain ha^{-1} was greater with LGP than with HGP. The coefficient of variation among the 28 seasons was slightly reduced with LGP (81%) compared with HGP (88%), indicative of only slight alleviation of

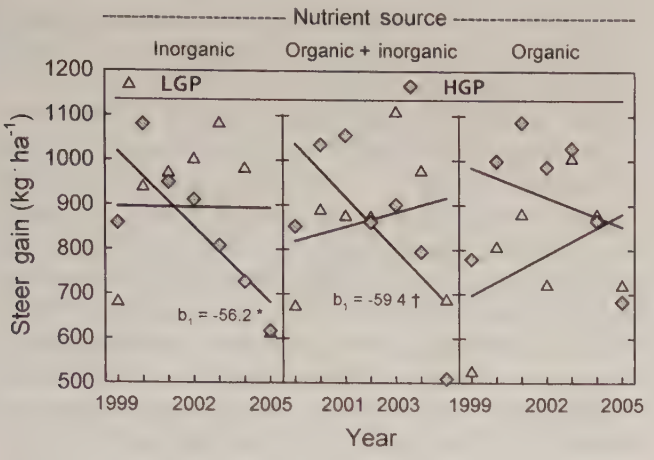

Figure 3 Relationship of time to steer gain ha^{-1} as affected by nutrient source and forage utilization regimes at Farmington, Georgia. † and * indicate significance of slope coefficient at $P=0.1$ and $P=0.05$, respectively. LGP, low grazing pressure; HGP. high grazing pressure.

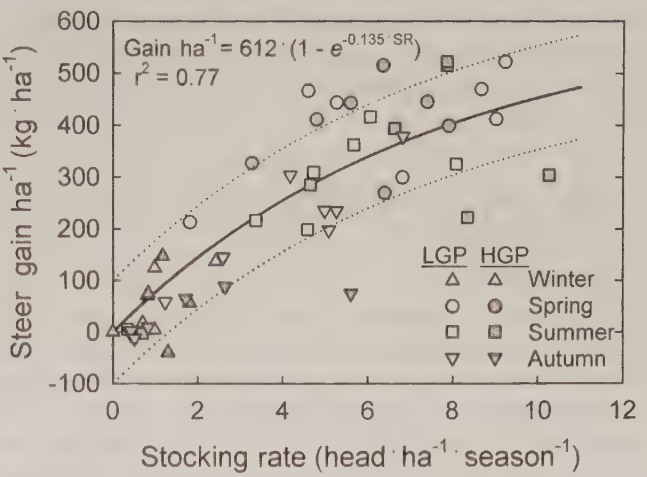

Figure 4. Relationship of steer gain ha^{-1} to stocking rate as affected by season and forage utilization regime at Farmington, Georgia. LGP, low grazing pressure; HGP, high grazing pressure. Dotted lines represent arbitrary boundaries of ±100 kg ha^{-1} from the regression relationship for illustrative purposes only.

environmental stress on steer production. The greater forage dry mass with LGP than with HGP allowed some buffering of variations in the environment on forage availability and intake.

Steer gain ha^{-1} was rather low, but surprisingly greater with LGP than with HGP in the autumn (134 versus 120 kg ha^{-1}, respectively; $P=0.10$) and in the winter (71 versus 44 kg ha^{-1}, respectively; $P=0.003$). It was not different between grazing pressure treatments in the spring (403 kg ha^{-1}). However, in the summer, steer gain ha^{-1} was lower with LGP than with HGP (260 versus 320 kg ha^{-1}, respectively; $P=0.002$). The summer response was associated with a significant nutrient source interaction with grazing pressure, in which steer gain ha^{-1} was not significantly different between grazing pressure treatments with inorganic fertilization (304 kg ha^{-1}), but was lower with LGP than with HGP under organic + inorganic fertilization (265 versus 316 kg ha^{-1}, respectively) and under organic fertilization (222 versus 336 kg ha^{-1}, respectively).

Over the entire 7-year period, steer gain ha^{-1} was not different between grazing pressure treatments when averaged across nutrient sources (863 kg ha^{-1}) (Table 5). However, there was lower gain ha^{-1} with LGP than with HGP under organic only fertilization (789 versus 918 kg ha^{-1}, respectively). Steer gain ha^{-1} was lower with LGP than with HGP in 1999, 2000 and 2001. There was no difference between treatments in 2002 but then there was a reversal of effects in 2003 and 2004 (and a similar trend in response in 2005). Steer gain ha^{-1} significantly declined with time under HGP when nutrient source was inorganic only or organic + inorganic fertilization, but was unchanged with time under LGP (Fig. 3). A difference in temporal trend with respective to forage utilization regime also occurred during Phase 1, but in contrast to increasing gain ha^{-1} under LGP and stable gain ha^{-1} under HGP in Phase 1[20], we observed stable gain ha^{-1} under LGP and declining gain ha^{-1} under HGP in Phase 2 (Fig. 3). The net difference between these two phases of the study was the same; although gain ha^{-1} became increasingly more improved with time under LGP relative to that under HGP. This differential response between grazing pressures with time suggests that accumulation of residual forage mass was better able to protect the pasture production system from the vagaries of weather-related and other environmental stresses.

Compared with Phase 1 of this experiment, steer gain ha^{-1} improved by 51% under LGP and 22% under HGP[20] (Table 5). Greater animal production was a result of more grazing days, despite the lower animal performance during the cooler portions of the year. Considering only the spring and summer grazing seasons in Phase 2, steer gain ha^{-1} was 666 kg ha^{-1} under LGP and 722 kg ha^{-1} under HGP, values that were similar to those in Phase 1 (578 and 726 kg ha^{-1} under LGP and HGP, respectively). Therefore, introduction of tall fescue into bermudagrass sod did not adversely affect animal production during the spring and summer, but rather simply added to animal production during the remainder of the year.

Considering the period of 2001–2004 when more complete full-year grazing was logistically possible, steer gain ha^{-1} averaged 930 kg ha^{-1}, a value similar to mean steer gain of 970 kg ha^{-1} from southern Georgia on different improved bermudagrass cultivars[25]. The high gain ha^{-1} from mixed tall fescue–bermudagrass pastures is encouraging, given that the cool-season forage was ergot-alkaloid producing. Other mixed-forage pastures may be potentially even more productive, as well as environmentally sustainable with regard to restoration of soil organic matter[26]. Further research on alternative

forages should be conducted in a long-term, holistic experimental setting, since time and multidisciplinary interactions were shown to be important in the current experiment as well.

Relationships among response variables and general discussion

Relationships among stocking weight, stocking rate, average daily gain and gain ha^{-1} were all positively significant ($n=56$; averaged across nutrient sources as in Fig. 2). Strongest relationships were between stocking rate and steer gain ha^{-1} (Fig. 4). Stocking rate varied due to availability of forage at the time of stocking and the projection for continued forage growth during the remainder of the stocking period. Steer gain ha^{-1}, therefore, became highly dependent on stocking rate. Steer performance was a secondary variable that affected the scatter in the relationship between stocking rate and gain ha^{-1}. There were some key outlying points in the relationship that indicated differences in how average daily gain might have played a role. Four steer gain ha^{-1} observations fell 100 kg ha^{-1} below the regression relationship, all of which were under HGP and two of which were in summer (2003 and 2005) and one each in autumn (2004) and winter (2005). All of these periods (except winter 2005) had low average daily gain (0.28 ± 0.07 kg d^{-1}) and relatively high stocking rate (8.1 ± 2.3 steers ha^{-1}), suggestive that poor performance was likely associated with significant ergot alkaloid consumption from abundant forage. Residual forage dry mass at the end of those periods was 1.5 ± 0.2 Mg ha^{-1}, a level that likely exposed a high proportion of tall fescue forage for consumption compared with bermudagrass. Winter 2005 had low average daily gain (-0.35 kg d^{-1}) and low stocking rate (1.3 steers ha^{-1}), and therefore, the lack of fit with the relationship was likely due to both ergot-alkaloid containing forage and low available forage.

In contrast, almost all of the steer gain ha^{-1} observations that were >100 kg ha^{-1} above the regression relationship in Figure 4 occurred during the spring ($n=6$) or summer ($n=2$). Most of these observations ($n=6$) were also under HGP. These results suggest that the spring and early summer period was not necessarily a period of reduced animal performance, since all of these periods had high average daily gain (0.89 ± 0.15 kg d^{-1}) and moderate stocking rate (5.7 ± 1.6 steers ha^{-1}). These observations are in contrast to previous literature, which indicates the greatest period of depressed animal performance from consuming ergot-alkaloid-containing tall fescue is in the spring and autumn [5,27,28]. It is possible that dilution of late-spring tall fescue forage consumption with emergent bermudagrass forage may have overcome any ergot-alkaloid consumption issue. In addition, maintaining low forage dry mass of tall fescue during the spring has been hypothesized as a strategy to minimize toxicity (J.A. Stuedemann, unpublished data). Residual forage dry mass was 1.3 ± 0.7 Mg ha^{-1} among these eight periods with high average daily gain.

Wide variations in seasonal values of animal responses (i.e., stocking density, stocking rate, average daily gain and gain ha^{-1}) illustrate the need for graziers to anticipate changes in seasonal weather conditions (Fig. 2). If rainfall and temperature could have been anticipated better, it is possible that a smoother temporal dynamic might have resulted. However, this dynamic also shows the difficulty in closely managing pastures to preset targets. These data indicate that good times on pasture are often followed by hard times and vice versa. Therefore, a conservative approach to stocking could help moderate extreme pasture conditions. In contrast, reactionary stocking to take advantage of available forage likely led to consumption of higher quality forage, as documented in other studies[14,29]. The put-and-take stocking system employed in this study can therefore be considered a hybrid stocking method between continuous and rotational stocking. Further research is needed to understand whether the dynamic pasture responses reported here might be found in more traditionally defined continuous stocking or rotationally grazed pastures.

Mixed-species composition of pastures was considered to be a more sustainable pasture system to allow nutrients to be used more effectively by actively growing forage throughout the year, rather than potentially lost during dormancy. In addition, filling dormant periods with a desired perennial forage species reduces the opportunity for weeds to develop and reduces resource requirements (e.g. labor, fuel and time for overseeding of annual forages). We have been measuring environmental indicators in this study, but they will be reported elsewhere.

The high rates of nutrient sources applied (285 ± 45 kg N ha^{-1} yr^{-1} as inorganic, organic+inorganic and organic) were likely key contributors to the excellent forage and steer production in this study. The amount of N harvested from grazed pastures can be assumed to be 3% of live-weight gain[30]. Therefore, N removal with steer production would have been only 26 ± 5 kg N ha^{-1} yr^{-1}. We also measured total soil N accumulation in the 0–6-cm depth of soil in these pastures as 74 ± 5 kg N ha^{-1} yr^{-1} [26] and accumulation in the 0–60-cm depth of soil as 239 ± 27 kg N ha^{-1} yr^{-1} [10]. Therefore, at least during the first 12 years of this experiment (Phases 1 and 2), most of the N applied to the pasture could be accounted for as harvested in cattle gain and stored in soil organic matter. There is the possibility that total soil N accumulation included a significant fraction of inorganic N that would be potentially mobile to the groundwater if leached or denitrified by soil bacteria *in situ*. Soil-profile inorganic N dynamics will be reported elsewhere. Rate of N application to well-developed pastures should decline with time to coincide with slower soil organic matter accumulation rates.

Conclusions

Stocker grazing of mixed tall fescue–bermudagrass pastures in the Piedmont of Georgia was highly successful in achieving adequate steer performance (0.61 ± 0.12 kg d^{-1}, mean ± standard deviation among 42 nutrient source × forage utilization × year combinations) and steer production (863 ± 157 kg ha^{-1}). Reducing grazing pressure to increase ($P < 0.001$) residual forage dry mass from 1.46 ± 0.29 Mg ha^{-1} (HGP) to 1.97 ± 0.32 Mg ha^{-1} (LGP) resulted in a reduction ($P < 0.001$) in steer stocking rate from 4.2 ± 0.8 steers ha^{-1} (HGP) to 3.7 ± 1.1 steers ha^{-1} (LGP). However, the reduction in grazing pressure was able to increase ($P < 0.001$) steer performance from 0.58 ± 0.11 kg d^{-1} (HGP) to 0.64 ± 0.12 kg d^{-1} (LGP), which resulted in nearly equivalent ($P = 0.24$) steer production of 876 ± 156 kg ha^{-1} (HGP) and 850 ± 160 kg ha^{-1} (LGP).

Nutrient source regime had little effect on residual forage mass level and on steer stocking rate and, therefore, broiler litter fertilization could be considered as effective as inorganic fertilizer in supplying sufficient nutrients for forage growth. However, the responses of steer performance and steer gain ha^{-1} were sensitive to nutrient source. Average daily gain under LGP was greater than under HGP with inorganic only and organic + inorganic fertilization, but was similar between grazing pressures under organic fertilization. This translated into no difference in gain ha^{-1} between grazing pressures with inorganic only and organic + inorganic fertilization, but lower gain ha^{-1} under LGP than under HGP with organic fertilization. The reason for this reduction in steer performance and production with organic fertilization under LGP could be linked to ingestion of a greater proportion of ergot-alkaloid-containing tall fescue forage or more potent concentration of such toxins in association with high broiler litter application.

The temporal shift from lower steer production with LGP than with HGP early in the study toward greater steer production with LGP than with HGP later in the study was a significant outcome of this study. This result suggests that more moderate grazing pressure in a tall fescue–bermudagrass pasture system will improve long-term sustainability, by allowing a greater buffer of residual forage mass to overcome environmental stresses caused by periods of short-term drought, as well as a more robust root system to absorb nutrients in the soil profile.

Resource efficiency of the tall fescue–bermudagrass pastures evaluated in this study was considered high for a number of reasons. First, utilization of broiler litter as a nutrient source utilized a locally abundant resource. The concerns with nutrient imbalance and potential water quality deterioration with repeated broiler litter still needs to be assessed in comparison with mixed inorganic and organic sources. Second, a nearly year-round grazing system was possible with mixed tall fescue–bermudagrass pastures to utilize land resources efficiently, as well as to improve the quality of degraded cropland. Third, targeted anthelmintic treatment of steers in spring and autumn reduced reliance on therapeutic drug treatment and kept pastures free of gastro-intestinal parasites.

We conclude that excellent cattle performance and productivity can be achieved with broiler litter fertilization of strategically grazed pastures with mixed tall fescue–bermudagrass forage composition. Combined with results from botanical composition changes during this experiment [22], a strategically grazed system would rely on closely grazing the robust tall fescue component during late winter to early spring and moderately grazing the bermudagrass component during the summer and autumn. These results can be used by producers to improve the sustainability of cattle grazing systems in warm, humid climates.

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Soil fertility and crop growth under poultry/crop integration

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Abstract

Innovative sources of soil fertility are of utmost importance to growers in light of rising fertilizer costs and environmental concerns. Integrating livestock and crop production is one channel by which agricultural practitioners can enhance soil fertility. For this research, soil fertility was analyzed in pastured poultry/crop agroecosystems to determine whether free-ranging birds and pasture could be used to replace or supplement non-manure-based fertilizers. Soils from adjacent cropped areas were compared to plots with a recent history of pastured poultry use on two farms, and crop plants were grown in each type of soil in a replicated greenhouse experiment. Spatial variation in soil fertility was also assessed relative to location of poultry coops. Pastured poultry plots had elevated soil total C, total N, NH_4^+–N, NO_3^-–N, Olsen P, exchangeable K, organic matter, cation exchange capacity and electrical conductivity relative to the control of typically managed organic farm soils without pastured animal inputs. These soil fertility changes conferred greater biomass and height to sunflowers and beans grown in these soils relative to control soils for most treatments. Results suggest that pastured poultry can effectively fertilize soil for certain crops but that a need exists for more research into (1) phosphorus management and (2) ensuring a spatially uniform distribution of manure.

Key words: crop/livestock systems, integrated agriculture, manure, pasture

Introduction

As the price of conventional fertilizer soars and interest in alternative sources of crop fertilizer grows[1], there is an increasing demand for on-farm sources of soil fertility[2]. Integrating livestock and crop production is one channel by which agricultural practitioners can enhance soil fertility using on-farm resources of animal manure and pasture plants[3–5]. Farmer interest in utilizing pasture-raised chickens (*Gallus gallus domesticus*), in particular, to improve soil fertility and enhance crop performance has grown in recent years, as has consumer demand for pastured poultry products[6–8]. In pastured production, poultry live outdoors and consume pasture plants and grain[9,10]. Relatively few peer-reviewed articles have been published regarding soil fertility and crop growth under pastured poultry management, yet inferences about the role of poultry on soil fertility can be made from the substantial literature on the fertilizer content values of poultry manure and litter[1,11–13].

Poultry manure and litter are known effective plant fertilizers. Various estimates give the nutrient concentrations of poultry manure at an average of 5.1% nitrogen (N), 1.9% phosphorus (P) and 2.6% potassium (K)[14–16]. Many field studies suggest that poultry manure and/or litter application to crop soils can enhance chemical, biological and physical quality, and plant growth. Sainju et al.[11] found broiler litter increased soil carbon (C) and N sequestration in intensive cropping systems at a depth of 0–10 cm. Adeli et al.[1] reported that broiler litter application to cropping systems linearly increased extractable soil P, K, soil cation exchange capacity and soil total C at 0–15 cm in a 3-year study. Land application of poultry litter has also been found comparable to inorganic fertilizer for supplying soil P and increasing forage yield[17].

Poultry manure and litter have been widely studied because the poultry industry generates high quantities that are frequently applied to crop or pasture fields[11,18]. Less is known about the effect of utilizing live birds for soil fertilization. Manure deposited by live birds is surface-applied and not tractor-incorporated; although Kratz et al.[19] reported that fecal N and P inputs by free-ranging birds corresponded with increased soil N and P levels. Studies such as these suggest that integrating live poultry into crop agroecosystems may effect significant changes to soil fertility and yield.

The objectives of this study were to: (1) determine whether soil from poultry pasture (PPAS) enhanced soil

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fertility in comparison with typical organic non-animal based management (control); and (2) identify whether crops grown in soil from PPAS fields achieved equivalent growth parameters as crops grown in control soils. These effects were assessed for two agroecosystems: (1) a mixed annual/berry system where crops were planted subsequent to removal of birds from pasture; and (2) an orchard setting where birds were temporally and spatially integrated with crop production.

Methods

Study region and experiments conducted

Research was conducted between 2009 and 2011 in the Central Coast region of California at two working farms. The region is characterized by a nearly year-round agricultural growing season with a mild Mediterranean climate. The rainy, cooler season is from November to April, and the dry, warm season is from May to October. Three assessments were conducted: for objective 1, soil fertility parameters were compared from adjacent plots on two farms, and spatial variation in soil fertility was assessed relative to location of poultry coops. For objective 2, crop plants were grown in a replicated greenhouse experiment using soils from plots with a recent history of pastured poultry and soils from adjacent cropped areas without a history of pastured poultry.

Site selection

This research was conducted on working farms, making site selection a critical aspect of experimental design. The advantage of agroecological studies based on working farms is that they reflect real-world farming conditions and generate data based on actual management practices. The disadvantage is a reduced ability to replicate treatments and choose ideal controls. In choosing sites for this research, we sought farms with management practices representative of integrated pastured poultry/crop production. Field histories and farmer interviews were used to determine that there were no *a priori* reasons for poultry placement related to soil fertility and that plots were similar enough to deduce whether differences were due to the presence or absence of poultry and pasture. Two farms were selected that met all requirements. At each farm we established two adjacent plots, one with a history of crop production and one with a history of pastured poultry. Adjacent plots had similar cropping histories and identical soil types and slopes.

Research sites

Farm A was used as an exemplar of PPAS in a mixed annual/berry agroecosystem. Soil is a Lockwood series (fine-loamy, mixed, Thermic and Pachic Argixerolls). Texture of the topsoil is clay (sand: 28.4%, silt: 28.4% and clay: 43.2%). At this site, hens were the integrated animals, and plots were alternately dedicated to crop and pasture production. Plot sizes were 24.5 m × 7 m (Fig. 1).

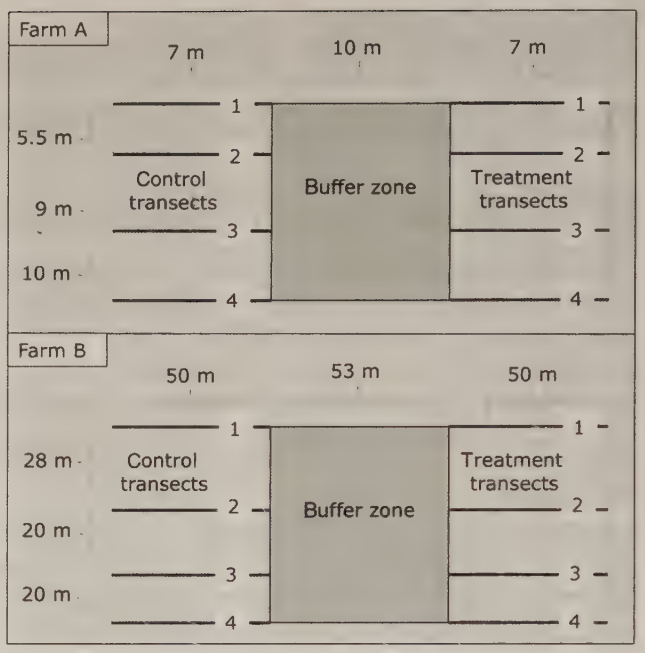

Figure 1. Experimental design of soil fertility study showing transects (bold lines), distance between transects, and buffer zones at both sites.

At this site, the pasture plot had a 2-year history of poultry grazing (2470 birds per ha). Pasture fields were seeded with fescue (*Festuca arundinacea*), orchardgrass (*Dactylis glomerata*), ryegrass (*Lolium* spp.), clover (*Trifolium* spp.) and trefoil (*Lotus corniculatus*), and intermittently ranged by hens for 2 years before being converted back to crop production. Prior to pasture, the treatment field was planted with popcorn (*Zea mays averta*) in 2007 and pumpkins (*Cucurbita* spp.) in 2006. During crop sequences, plots were planted with cover crops [bell beans (*Vicia faba*), purple vetch (*Vicia benghalensis*), peas (*Pisum sativum*) and oats (*Avena sativa*)] during the winter season. While managed as pasture, overhead irrigation was used, and during crop rotations, drip irrigation was employed.

The control field had a 2-year history of wheat (*Triticum* sp.) and popcorn (*Zea mays averta*) during the time the treatment field was planted as pasture and grazed by poultry. Previously, the control field was planted with wheat in 2007 and popcorn in 2006. Winter cover crops, as described above, were planted into these crop rotations. Compost and lime were added to all crop sequences biennially, both at a rate of 2.4 Mg ha^{-1}. Both fields were converted to strawberry (*Fragaria* sp.) production subsequent to the first day of data collection. During strawberry cultivation, winter cover crops were not utilized.

Farm B was used as an exemplar of PPAS in an orchard system. Soil is a Sorrento series (fine-loamy, mixed, Thermic and Calcic Haploxerolls). Texture of the topsoil is sandy clay loam (sand: 51.4%, silt: 26.4%, and clay: 22.2%). At this site, meat chickens were integrated into a portion of walnut (*Juglans regia*) orchard on pasture seeded as above. Plot size was 68 m × 50 m (Fig. 1). An equally sized section of the orchard with no prior history of poultry was used as the control, and both plots had equivalent walnut tree density (100 trees per ha). Chickens grazed the PPAS orchard from April to September annually for the previous 7 years (2470 birds per ha). The control orchard had no history of poultry and was planted with the same cover crop mix described above. No other amendments were added to either plot. Both plots were irrigated on the same schedule, receiving water three times per year for 12 h.

Data collection for soil fertility

This assessment was conducted from 2009 to 2011. Using a stratified composite sampling design[20], eight composite samples per plot were gathered. Each composite sample was composed of 15 thoroughly combined sub-samples. Samples were gathered from two depths (0–15 and 15–30 cm) along eight permanent transects per site in two plots (Fig. 1), and sub-samples of each composite sample were used for analysis. Field history and farmer interviews were obtained to determine that no *a priori* reasons existed for delineation of field margins (e.g., soil-type differences or productivity), and transects were placed so that the distance between transects was greater than the distance between treatments to rule out stochastic spatial variation. At Farm A, transects were not true replicates because the treatment field was grazed by a single flock of hens, meaning that all data collected represent the effect of one flock. For this reason, we present data from this site as repeated measures. At Farm B, transects were true replicates because meat chickens were housed in discrete pens, and each transect represents the effect of a distinct flock. Data from this site are presented as means across time.

Inorganic N was measured four times per year ($n = 256$)[21], analyzed for ammonium–nitrogen (NH_4^+–N) and nitrate–nitrogen (NO_3^-–N) on a flow injection analyzer (Lachat QuikChem 8000, Milwaukee, WI)[22] at the University of California, Santa Cruz (UCSC). Total C and total N were measured twice yearly ($n = 64$) and assessed on the VarioMAX CNS analyzer (Elementar, Mt. Laurel, NJ)[23] at UCSC. Calcic properties of Farm B soils were limited to >50 cm depth, making inorganic C removal from soils prior to total C analysis unnecessary because sample soils were collected from surface depth increments of 0–15 and 15–30 cm. Soil electrical conductivity[24] was determined twice yearly ($n = 31$, missing one sample, topsoil only) as a saturated paste using a conductivity meter (Denver Instrument, Model 250) at UCSC. Extractable sodium bicarbonate P (Olsen P), exchangeable K (ex. K), pH (saturated paste), cation exchange capacity and soil organic matter as determined by weight loss-on-ignition were assessed four times per year ($n = 128$)[25] at A&L Laboratories (Modesto, CA).

In a sub-study conducted in February 2010, spatial variation in soil fertility was assessed for the coop-based system at Farm A. Hens, unlike the meat chickens at Farm B, ranged within their fields during the day, frequently seeking shelter under the coop. This study was conducted to determine whether soil fertility values changed with distance from the coop. Samples were taken every 4 m on four 16-m transects at two coops ($n = 20$). Soil samples were collected from a depth of 0–15 cm and analyzed for inorganic N, Olsen P, ex. K, pH, cation exchange capacity and organic matter, using the methods described above.

Greenhouse trial

To address the question of crop growth response to PPAS independent of farmer management, a series of greenhouse experiments were conducted using soil from PPAS and control plots. From each plot, five gallons of topsoil (0–15 cm depth) were collected from 20 random sub-samples. Soil was then air-dried, mixed well and crushed to approximately 2 mm diameter aggregates. Two bean seeds (*Phaseolus vulgaris* L. var. Hystyle) were planted on November 5, 2009 with Farm A soil and thinned to one shoot on November 11, 2009 ($n = 33$). In the same manner, sunflowers (*Helianthus annuus* L.) were planted on April 23, 2010 with soil from Farms A and B and thinned on April 29, 2010 ($n = 174$). Containers were arranged in a random formation, misted daily and grown at daytime temperatures of 22.8–23.9°C for 30 days in the UCSC greenhouse. At 30 days from seeding, plants were clipped at the soil level and measured for height. Plants were then oven-dried at 60°C for 36 h and the biomass weighed.

Data analysis

True replication of treatment within site at Farm A was not viable due to the scale of the treatment, and management practices between the two farms differed too greatly to warrant statistical comparison across farms. Instead, we present results as two distinct case studies. Two-tailed Student's *t*-tests were conducted for most data analysis with the exception of the spatial variation study, where simple linear regressions were used to determine the effect of distance from coop on soil fertility parameters. JMP, version 8 was used for all statistical tests[26] with the alpha level set at 0.05. The following data with non-normal distributions were transformed to meet assumptions of normality: $\log_{10}$ of total C at Farm A (15–30 cm) and Farm B (both depths); total N at Farm B (15–30 cm); NH_4^+–N at Farm A (15–30 cm) and Farm B (both depths); NO_3^-–N for all tests; Olsen P at Farm A (0–15 cm) and Farm B (both depths); ex. K at Farm B (15–30 cm); and

Table 1. Farm A soil fertility results for the 0–15 cm depth except where indicated.

Parameter	Days since birds removed	Mean PPAS	Mean control	df	*t*-stat.	*P*-value	dir.
pH	0	6.90 (0.09)	6.55 (0.09)	5.98	2.78	**0.0320***	+
	98	7.18 (0.05)	6.88 (0.09)	4.72	3.06	**0.0302***	+
	181	7.03 (0.05)	6.85 (0.05)	5.99	2.53	**0.0500***	+
	384	7.25 (0.12)	7.30 (0.16)	5.57	0.25	0.8096	0
Total C ($g kg^{-1}$)	0	47.50 (1.28)	47.24 (1.23)	5.99	0.15	0.8889	0
Total N ($g kg^{-1}$)	0	4.13 (0.04)	3.96 (0.04)	5.89	3.01	**0.0241***	+
	441	4.16 (0.04)	4.04 (0.06)	4.95	1.58	0.1750	0
NH_4^+–N ($mg kg^{-1}$)	0	3.30 (1.20)	2.29 (0.41)	3.68	0.80	0.4713	0
	51	2.28 (0.61)	1.41 (0.15)	3.39	1.39	0.2500	0
	98	3.87 (0.35)	2.49 (0.24)	5.29	3.29	**0.0200***	+
	147	2.20 (0.09)	2.27 (0.41)	3.23	0.15	0.8894	0
NH_4^+–N ($mg kg^{-1}$)	0	3.40 (0.78)	2.23 (0.21)	3.81	1.36	0.2500	0
(15–30 cm)	51	2.38 (0.55)	0.90 (0.11)	4.85	3.50	**0.0182***	+
	98	4.60 (1.08)	3.47 (1.47)	4.83	1.04	0.3478	0
NO_3^-–N ($mg kg^{-1}$)	0	32.00 (7.81)	15.55 (1.60)	3.97	2.36	0.0778	0
NO_3^-–N ($mg kg^{-1}$)	0	26.91 (9.70)	15.67 (1.96)	3.16	0.04	0.9674	0
(15–30 cm)	51	40.60 (6.69)	20.74 (2.21)	5.11	3.23	**0.0211***	+
	98	11.58 (1.23)	24.20 (0.99)	3.85	6.50	**0.0033***	–
	147	32.37 (5.27)	39.89 (6.26)	5.99	0.83	0.4385	0
Olsen P ($mg kg^{-1}$)	0	29.90 (3.63)	24.25 (8.08)	4.21	1.02	0.3617	0
Ex. K ($mg kg^{-1}$)	0	552.25 (22.09)	561.53 (21.95)	6.00	0.30	0.7759	0
Organic matter ($g kg^{-1}$)	0	62.50 (0.87)	58.50 (0.96)	5.94	3.10	**0.0214***	+
	98	62.00 (0.58)	63.00 (1.41)	3.97	0.65	0.5487	0
Cation exchange capacity ($cmol kg^{-1}$)	0	25.10 (0.48)	26.15 (0.42)	5.91	1.65	0.1512	0
Electrical conductivity ($dS m^{-1}$)	0	0.72 (0.03)	0.50 (0.05)	4.87	8.57	**0.0004***	+
	98	0.53 (0.04)	0.52 (0.06)	4.79	0.43	0.6885	0

Figures in parentheses are standard errors.
Results are shown from time zero until the sampling date after which all differences were neutral.
'dir.' indicates direction of change for significant results of PPAS relative to the control.
* indicates statistically significant differences.

organic matter at Farm B (15–30 cm). For the spatial variation data, NH_4^+–N, NO_3^-–N and Olsen P were $\log_{10}$ transformed, and ex. K data were square root transformed. However, original means and standard errors are reported and graphically displayed hereafter for ease of interpretation.

Results

Effects on soil fertility

At Farm A, PPAS plots had elevated soil pH, total N, organic matter and electrical conductivity levels relative to the control at a depth of 0–15 cm at day 0, the day birds were removed from pasture and pre-conversion to strawberries (Table 1). Subsequent to removal of birds from pasture and conversion to strawberries, pH values remained significantly higher. Soil inorganic N differences were statistically significant for NH_4^+–N at 98 days after conversion at the 0–15 cm depth and at 51 days for the 15–30 cm depth. Soil NO_3^-–N differences differed significantly at 51 and 98 days after conversion at the 15–30 cm depth. No other significant differences were found at the 15–30 cm depth.

At Farm B, PPAS plots had decreased soil pH and elevated total C, total N, NH_4^+–N, NO_3^-–N, Olsen P, ex. K, organic matter, cation exchange capacity and electrical conductivity levels relative to the control (Table 2). These results were consistent at the 15–30 cm depth with the exception of soil organic matter.

Spatial variation in soil fertility results

Certain soil fertility parameters varied strongly along spatial gradients within PPAS fields where free-ranging hens lived. Regression analysis showed that soil NH_4^+–N, NO_3^-–N, Olsen P and ex. K all significantly decreased with distance from coop at a depth of 0–15 cm (Fig. 2). Soil pH, organic matter and cation exchange capacity were unaffected. Correlations of the following parameters and distance from coop were significant: NH_4^+–N ($r^2 = 0.42$, $F_{1,18} = 13.27$, $P = 0.0019$); NO_3^-–N ($r^2 = 0.79$, $F_{1,18} = 68.28$, $P < 0.0001$); Olsen P ($r^2 = 0.06$, $F_{1,18} = 1.15$, $P < 0.0001$); ex. K ($r^2 = 0.33$, $F_{1,18} = 8.88$, $P = 0.008$).

Greenhouse trial

PPAS soil improved crop growth relative to the control for both biomass and plant height, showing a neutral effect

Table 2. Farm B soil fertility results.

Parameter	Soil depth (cm)	Mean PPAS	Mean control	df	*t*-stat.	*P*-value	dir.
pH	0–15	6.99 (0.06)	7.36 (0.05)	27.59	4.96	**<0.0001***	–
	15–30	7.10 (0.06)	7.43 (0.05)	27.66	4.48	**0.0001***	–
Total C ($g\,kg^{-1}$)	0–15	16.07 (0.90)	11.26 (0.25)	7.48	5.35	**0.0009***	+
	15–30	22.7 (6.73)	7.86 (0.33)	6.28	3.22	**0.0169***	+
Total N ($g\,kg^{-1}$)	0–15	1.54 (0.10)	1.05 (0.00)	6.01	5.06	**0.0023***	+
	15–30	1.96 (0.52)	0.76 (0.03)	6.31	3.24	**0.0164***	+
NH_4^+–N ($mg\,kg^{-1}$)	0–15	4.97 (1.48)	1.24 (0.17)	51.76	4.32	**<0.0001***	+
	15–30	2.65 (0.57)	0.86 (0.16)	52.98	4.25	**<0.0001***	+
NO_3^-–N ($mg\,kg^{-1}$)	0–15	18.70 (3.04)	12.69 (2.70)	60.47	2.27	**0.0267***	+
	15–30	15.14 (2.18)	9.73 (1.68)	58.36	2.40	**0.0198***	+
Olsen P ($mg\,kg^{-1}$)	0–15	60.33 (5.14)	25.03 (2.38)	28.98	6.91	**<0.0001***	+
	15–30	44.72 (4.40)	13.81 (0.99)	28.33	10.11	**<0.0001***	+
Ex. K ($mg\,kg^{-1}$)	0–15	542.17 (17.21)	311.57 (12.57)	26.00	10.82	**<0.0001***	+
	15–30	391.5 (28.4)	224.53 (11.37)	23.98	6.13	**<0.0001***	+
Organic matter ($g\,kg^{-1}$)	0–15	28.60 (1.14)	22.63 (1.02)	28.42	3.90	**0.0003***	+
	15–30	22.53 (1.65)	19.44 (1.59)	28.22	1.70	0.0995	0
Cation exchange capacity ($cmol\,kg^{-1}$)	0–15	18.39 (0.22)	16.08 (0.50)	20.54	4.27	**0.0004***	+
	15–30	17.71 (0.62)	15.63 (0.34)	21.78	2.94	**0.0076***	+
Electrical conductivity ($dS\,m^{-1}$)	0–15	0.52 (0.04)	0.35 (0.02)	9.95	3.75	**0.0038***	+

Figures in parentheses are standard errors.
'dir.' indicates direction of change for significant results of PPAS relative to the control.
* indicates statistically significant differences.

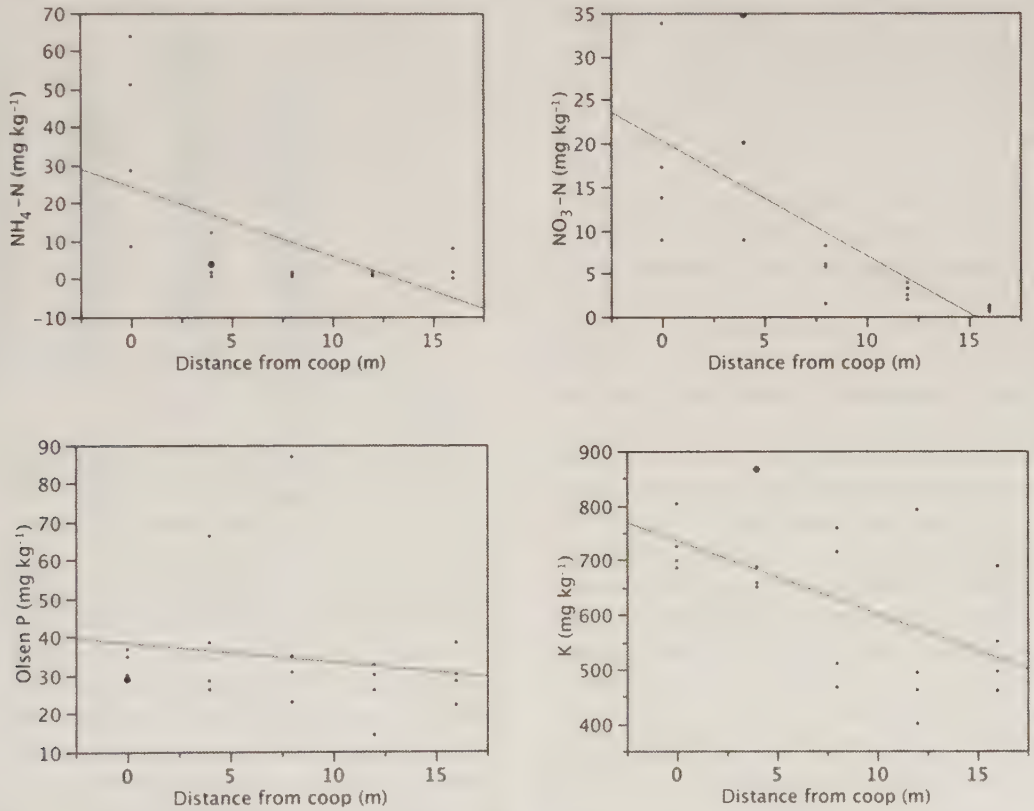

Figure 2. Regressions of soil ammonium–nitrogen (NH_4^+–N), nitrate–nitrogen (NO_3^-–N), extractable sodium bicarbonate P (Olsen P) and exchangeable K (ex. K) showing significant correlations to distance from coop.

only for sunflower biomass at one site. PPAS soil enhanced average bean biomass by a factor of 1.48 in Farm A soil (Fig. 3, $t(30.48)=3.31$, $P=0.0024$; Farm B soil not tested). Biomass results for sunflowers varied by farm. At Farm A, PPAS soil enhanced average sunflower biomass by a factor of 1.08 (Fig. 4, $t(54.94)=3.56$, $P=0.0008$). At Farm B, there was no significant difference in average biomass between plants

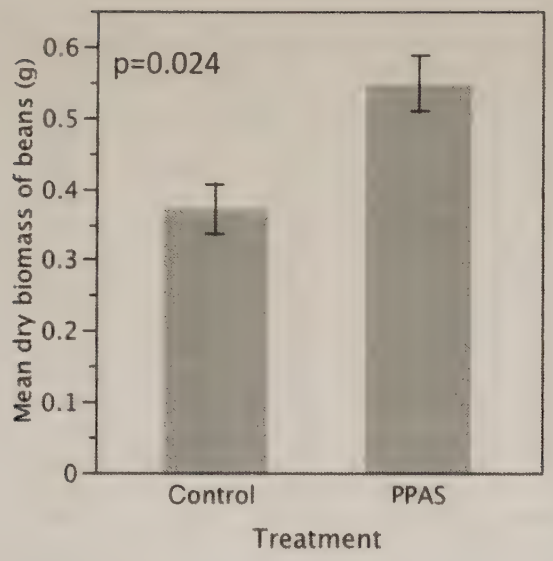

Figure 3. Biomass (grams/plant) of beans grown in soils from the control and PPAS with standard errors.

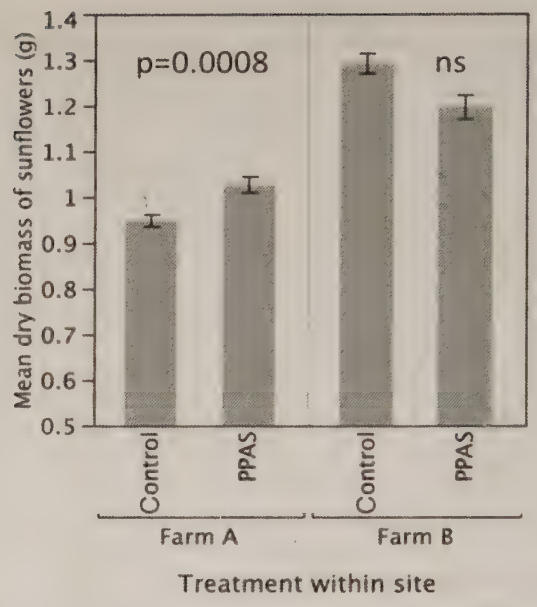

Figure 4. Biomass of sunflowers (grams/plant) grown in soils from the control and PPAS within farm with standard errors.

grown in PPAS and control soils (Fig. 4, $t(49.28) = 1.51$, $P = 0.1373$). PPAS soil enhanced sunflower height relative to the control group by a factor of 1.11 at Farm A (Fig. 5, $t(57.92) = 2.09$, $P = 0.041$) and by a factor of 1.06 at Farm B (Fig. 5, $t(108.17) = 2.30$, $P = 0.023$).

Discussion

PPAS plots had elevated soil fertility parameters of soil total C, total N, NH_4^+–N, NO_3^-–N, Olsen P, ex. K, organic matter and cation exchange capacity levels relative to the control. These changes conferred greater biomass and height to plants grown in these soils in the greenhouse in most cases. Some of these soil fertility parameters varied along spatial gradients within the PPAS fields where free-ranging hens lived. PPAS plots had highly increased soil Olsen P levels relative to the control at Farm B, with implications for environmental contamination of adjacent waterways. These results suggest that, despite the fact that in pasture-based systems manure is applied gradually by poultry rather than as a one-time mechanically incorporated application, PPAS does significantly affect soil fertility for multiple parameters, although it is not clear whether this effect is due to poultry manure or pasture plants.

Soil pH was raised relative to the control at Farm A, which is consistent with other studies of the effect of poultry manure and litter on soil acidity[1,27,28]. Hen poultry manure tends to increase soil pH because poultry feed contains calcium carbonate ($CaCO_3$) as an additive[29]. Feeds designated for hens contain more $CaCO_3$ than meat chicken feed to aid in shell production. Contrary to this phenomenon, manure in general can have a soil acidifying effect; when NH_4^+ is converted to NO_3^- and H^+ ions are released to the soil, pH is lowered[30].

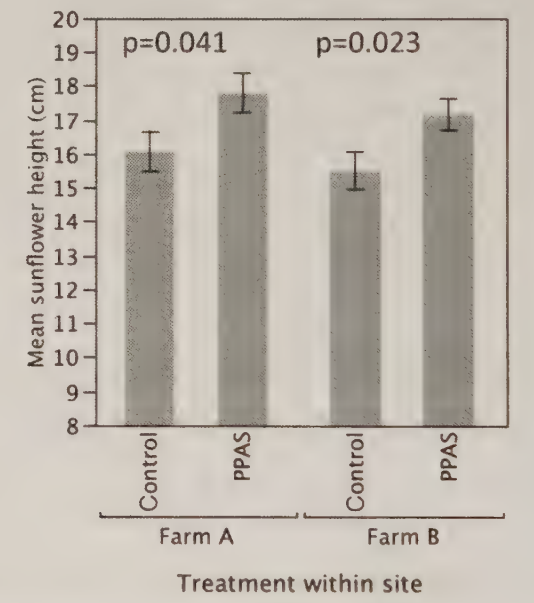

Figure 5. Height of sunflowers (cm/plant) grown in soils from the control and PPAS within farm with standard errors.

At Farm B, PPAS decreased soil pH. At this site, meat chickens were the integrated animal, which means that substantially less $CaCO_3$ was present in the feed, and the liming effect of the feed was apparently not greater than the acidifying effect of the manure. Also having an effect on soil pH at these sites was the buffering capacity of the soil, as determined by cation exchange capacity. At Farm A, soil cation exchange capacity was higher than Farm B soil cation exchange capacity, meaning that more $CaCO_3$ would be required to increase the pH of the soil than at Farm B. With hen feed containing more $CaCO_3$ than meat chicken feed, this effect was achieved. These

inorganic C additions were likely minor sources of increased total C measured in PPAS soils. Rather, soil total C increases are attributed largely to significant increases in organic matter.

PPAS plots showed elevated soil total C relative to the control at Farm B, the site with a 7-year history of PPAS. Similarly, other studies of manure have found a relationship between long-term manure applications and soil total C increase[1,31]. Soil total N was also increased by PPAS at both sites. Total N in poultry manure comprised 40–70% uric acid ($C_5H_4N_4O_3$), which is readily converted to ammonia (NH_3) in soil[19,32]. NH_3 is then either subject to volatilization or conversion to NH_4^+, depending on soil pH, and eventually NO_3^- by nitrification. NH_4^+–N and NO_3^-–N were increased in PPAS plots at both sites. Soil ex. K was also enhanced in PPAS plots at Farm B.

In addition to these chemical changes, soil organic matter was raised relative to the control in PPAS plots at the 0–15 cm depth for both sites. High soil organic matter is associated with high soil nutrient-holding capacity, low penetration resistance and a biologically active soil ecosystem[13,33,34]. Soil cation exchange capacity was also raised in PPAS plots relative to the control at Farm B but not Farm A, which is explained by the soil type of each site. Farm B had a sandy clay loam with 22% clay content, while Farm A had a clay soil with 43% clay content. At Farm A, the control soil had a substantially higher cation exchange capacity than the control Farm B, which is consistent with cation exchange properties of clay. At Farm B, PPAS plots has significantly elevated soil organic matter levels, which would lead to an increase in measureable cation exchange capacity for a soil not dominated by clay. Soil electrical conductivity was also increased in PPAS plots at both sites, although not beyond established threshold values for salt-sensitive crops[35].

Soil P levels were highly increased in PPAS plots relative to the control at Farm B. Phosphate is an additive to poultry feed, and excessive P accumulation associated with the use of poultry manure as a crop fertilizer is a well-known phenomenon[36–38]. Crops utilize approximately six times more N than P, yet N:P ratios are typically closer to 1:1 or 2:1 in manure[37]. Growers often apply manure to meet crop growth goals or N levels, resulting in P excess. Excessive soil P is lost primarily to overland runoff of P-enriched sediment and manure particles[39]. Loss of P to adjacent waterways can lead to eutrophication[40]. Whether soil P levels associated with pastured poultry pose an environmental risk is unique to each farm, as chemical and hydrological factors determine P retention and movement[41]. However, suggested threshold values of soil Olsen P for environmental quality are approximately 30–40 mg kg^{-1} [42,43]. Farm B levels exceeded this threshold for the PPAS treatment, and Farm A levels were close to the lower bound of this threshold in the PPAS treatment. Growers should be aware of this issue and conduct regular soil tests. When soil P reaches unacceptable levels for local conditions, birds should be relocated.

Finally, another issue of concern was the spatial variation in soil NH_4^+–N, NO_3^-–N, Olsen P and ex. K within PPAS fields where free-ranging hens lived. Hens tend to spend more time near or in the coop than toward the field margins, with the obvious result that they deposit more manure near the coop. Kratz et al.[19] found similar hot spots in free-range meat chicken runs. This issue can be ameliorated with more frequent relocation of portable coops. Research is required to determine how often coops should be moved to maximize nutrient deposition.

The relevance of these values to plant growth and yield depends on the type of crop grown with or subsequent to PPAS. In the case of this study, PPAS levels were at least as effective as the input methods growers were using for control soil preparation as shown by the growth results in the greenhouse trial. At the study sites reported on in this paper, the equivalent quantity of dry manure applied to a plot was approximately 8.4 Mg ha^{-1} yr^{-1} for layers and 10.9 Mg ha^{-1} yr^{-1} for broilers. Studies using comparable application rates have found that 10 Mg ha^{-1} of dry poultry manure resulted in higher N uptake by barley than inorganic N fertilizer alone at comparable rates of inorganic N application[44]; a litter application rate of 3.7 Mg ha^{-1} produced lint yields equivalent to maximum achievable with inorganic fertilizer, while 8.7 Mg ha^{-1} maximized lint yield[12]; and litter application for cotton at a rate of 6.7 Mg ha^{-1} maximized plant N utilization, while at rates of 10.1 and 13.4 Mg ha^{-1}, bulk density was improved relative to inorganic N treated soils[1].

Conclusions and future research

Innovative sources of soil fertility enhancement are of utmost importance to growers in light of rising fertilizer costs and environmental concerns. This study demonstrates the utility of pastured poultry as a technique for soil fertility enhancement on intensively managed organic farms. We found increased levels of soil total C, total N, NH_4^+–N, NO_3^-–N, ex. K, organic matter and cation exchange capacity in PPAS plots. These soil fertility changes conferred greater biomass and height to sunflowers and beans grown in these soils. This management practice is not without drawbacks, however, and future research is necessary to mitigate nutrient management issues.

Particular attention is needed regarding soil N and P dynamics. This study suggests that 7 years of PPAS leads to excessively high Olsen P levels (Farm B). Research should be directed at (1) determining the amount of time birds should be withheld from a field before reintroducing a pasture rotation; and/or (2) a stocking density that optimizes soil P levels. At optimal P levels, N levels will likely be inadequate. To add more N, leguminous winter cover crops could be used. N-scavenging cover crops could also be used to minimize N loss and to recycle

inorganic N. Radish (*Raphanus sativus*), mustard (*Brassica hirta* and *Brassica alba*), phacelia (*Phacelia tanacetifolia*), rye (*Secale cereale*) and annual ryegrass (*Lolium multiflorum*) are all known nitrogen scavengers[45,46] that could function to recycle N from poultry manure. Finally, altering supplements added to poultry feed may impact water-soluble soil P levels and is an area of ongoing research[47]. Movement of birds within a field also needs to be researched in order to optimize nutrient management. Hot spots can be addressed by determining how long a coop can remain in one location before leading to excessive nutrient accumulations.

Investigations into nutrient management and hot spots are imminent needs of future research that will directly impact management. Additional areas of research will serve to elucidate mechanism, adding to a theoretical understanding of integrated crop/livestock systems, and potentially influencing management. This research does not distinguish between the roles of manure and pasture plants in determining soil fertility changes. Although poultry manure undoubtedly determines certain soil-quality changes in pastured poultry/crop systems, pasture plants are also likely to play a role. Maintaining a field with a pasture mix of grass and legumes is functionally similar to cultivating a cover crop[48], and the effects of cover crops on nutrient cycling, organic matter and physical soil quality are well known[2,49]. Additionally, pasture is tilled less frequently than crop fields, potentially leading to decreased erosion and higher carbon storage[50,51]. In order to develop a mechanistic understanding of the relative contribution of manure versus pasture plants on soil quality, further research utilizing chicken exclosures is necessary.

Integrating pastured livestock with crop production is an emerging phenomenon on the Central Coast of California. Our study demonstrates that integrated pastured poultry systems appear to be an effective practice for managing soil fertility and crop growth. When managed with care, integrated pastured poultry/crop agroecosystems have the potential to contribute to the sustainable production of food via recycling of manure, bolstering of soil fertility and reduction of purchased fertility inputs.

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Long-term farming systems research in the central High Plains

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Abstract

In recent decades, there has been growing interest among farming and scientific communities toward integrated crop–range–livestock farming because of evidence of increased crop production, soil health, environmental services and resilience to increased climatic variability. This paper reviews studies on existing cropping systems and integrated crop–range–livestock systems across the USA which are relevant in the context of summarizing opportunities and challenges associated with implementing long-term crop–range–livestock systems research in the highly variable environment of the central High Plains. With precipitation ranging from 305 to 484 mm and uncertain irrigation water supply, this region is especially vulnerable to changing moisture and temperature patterns. The results of our review indicate that diverse crop rotations, reduced soil disturbance and integrated crop–livestock systems could increase economic returns and agroecosystem resilience. Integrating agricultural system components to acquire unique benefits from small- to medium-sized operations, however, is a challenging task. This is because assessment and identification of suitable farming systems, selection of the most efficient integration scheme, and pinpointing the best management practices are crucial for successful integration of components. Effective integration requires development of evaluation criteria that incorporate the efficiency of approaches under consideration and their interactions. Therefore, establishing the basis for more sustainable farming systems in the central High Plains relies on both long-term agricultural systems research and evaluation of short-term dynamics of individual components.

Key words: central High Plains, crop–range–livestock system, interdisciplinary long-term experiments, production approaches, irrigated cropping systems

Introduction

Escalating prices of agricultural commodities and increasing climatic variability are pressing producers to adopt agricultural systems that meet food and fiber requirements and are economically and ecologically resilient[1]. Semi-arid agro-ecosystems near the lower limit of rain-fed crop production are especially vulnerable to changes in climate, input costs and prices, yet they cover vast areas of the western USA and the world. Integrated crop–livestock farming can optimize use of available resources by increasing the diversity of crops grown, while minimizing economic risks and negative environmental effects of more specialized systems[2]. In many other regions, long-term systems-focused research shows positive results from diverse, integrated farming systems that are risk-averse and resilient[3]. Therefore, whole-system, integrated research may become increasingly valuable to facilitate adaptation to changing climatic and economic conditions in the landscapes of mixed rangelands, crops and forages of the central High Plains, including parts of Wyoming, Nebraska and Colorado[4] (Fig. 1).

Long-term agricultural systems research can reveal relationships among economic, social and biological parameters at the core of landscape-scale environmental problems, such as erosion, pollution, and habitat loss[5], while also indicating economic viability of alternative, lower impact or value-added production approaches. To accomplish this, systems research requires broad frames of reference that integrate crop, range, livestock and other production components over at least one rotation period. But the most meaningful information about agricultural system resiliency and economic viability comes from very

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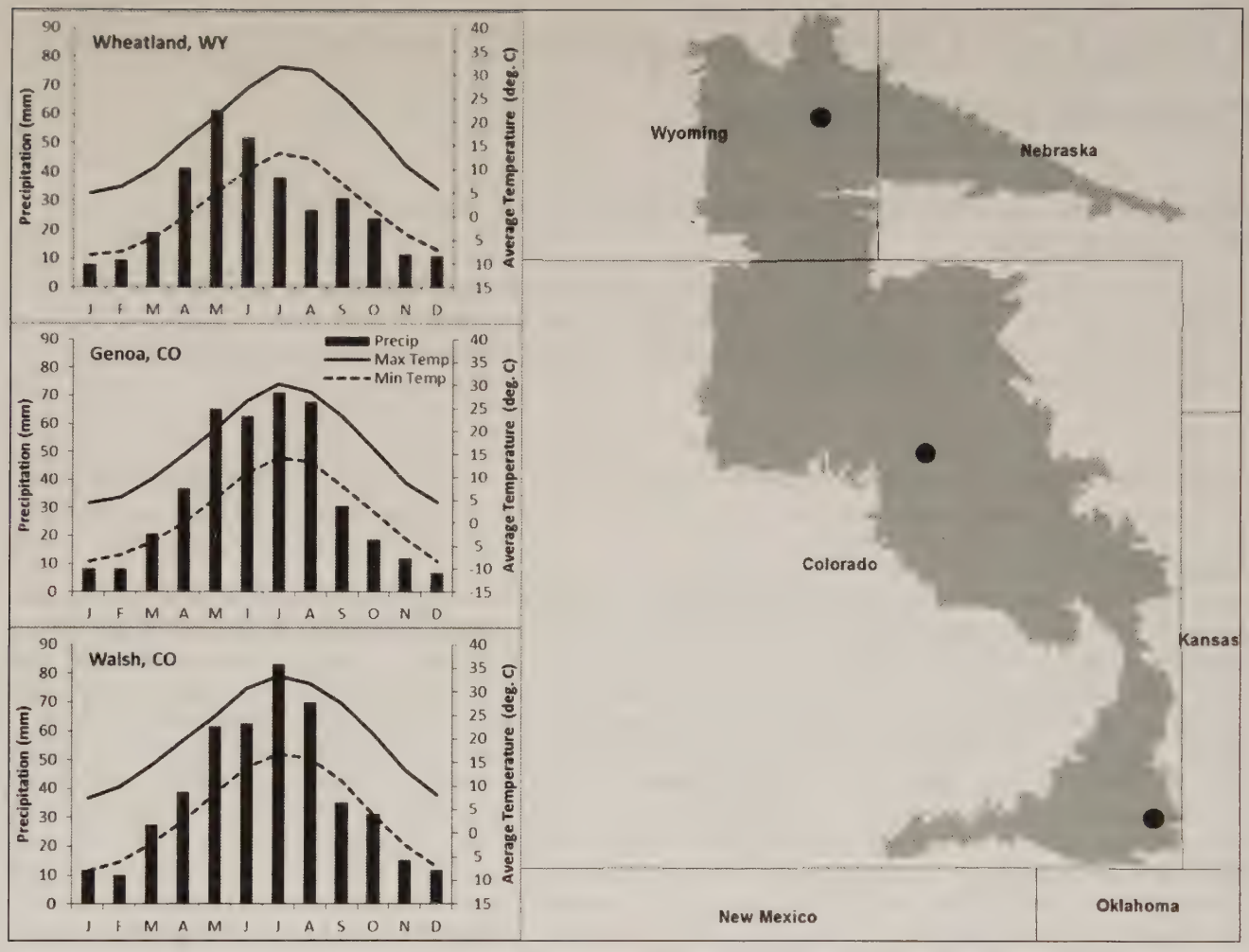

Figure 1. Long-term average temperature (lines) and precipitation (bars) in the central High Plains[4,6].

long-term projects in which technologies change to reflect changing production practices, and in which short-term research on particular innovations is often embedded. Moreover, such long-term systems research projects provide excellent settings for integrating multiple scientific objectives with education, and extension that truly engages producers, researchers and educators[3].

As concerns about a globalizing food economy, a warming climate, emerging pest and disease problems, human health and the environment, and the quality of life in rural areas converge[7], a consistent, multivariate evaluation system that makes the most of long-term research becomes ever more crucial. Although multidisciplinary in approach, the results of systems research are not often evaluated in terms of economic and ecological efficiency and interrelationships among system components[8] as individual researchers rush to publish results in their own disciplines. Our objective for this review was to develop an inventory of knowledge about agricultural systems management gained from long-term, integrated research, especially as it pertains to three broad production approaches toward management of crop–range–livestock systems in the central High Plains Major Land Resource Area[4]: conventional or typical management, reduced-input management, and certified organic management. We discuss the need, not only for interdisciplinary project implementation and data collection but also for integrated, multivariate data analysis so that system-level conclusions can be drawn about sustainability and economic profitability.

Study Approach

To establish an understanding of knowledge gained from systems-level agricultural research, we reviewed the literature in peer-reviewed scientific journals reporting results from long-term cropping systems or integrated crop–range–livestock systems projects. We used key words emphasizing components of conventional, reduced-input and certified organic production, long-term farming systems in the USA, integrated crop–range–livestock systems, central High Plains agriculture, agricultural system experiments, cropping systems studies in the central High Plains, and crop and livestock components in both irrigated and dryland conditions.

Agroecosystems of the central High Plains represent a broad swath of the semi-arid Western USA, generally too dry and cold for cultivation of many crops (Fig. 1). The region is suitable for integrated crop–range–livestock systems, often including irrigated rotations of alfalfa (*Medicago sativa* L.) and commodity crops such as corn (*Zea mays* L.), dry beans (*Phaseolus vulgaris* L.), sugar beets (*Beta vulgaris* L.), and others, or dryland rotations centered on winter wheat (*Triticum aestivum* L.) and often including millet (*Panicum* spp.), sunflowers (*Helianthus annuus* L.), corn and other grain crops; and rangelands for summer forage.

Under conventional management typical in this region, producers use inputs and technology as needed to maximize production. They rely chiefly on synthetic fertilizers for soil-fertility renewal and on combinations of herbicides and tillage for weed control. For this paper, the reduced-input approach is defined as management that offsets fertilizer use by building soil organic matter (SOM), mainly through reduced disturbance. Weed control in the reduced-input approach relies mostly on herbicides in conjunction with herbicide-resistant crop varieties. In the organic production approach, soil, crop and livestock management are based on practices allowed by the United States Department of Agriculture (USDA) National Organic Program standards. Soil-fertility renewal relies on inputs of organic materials along with legumes and cover crops in rotation. Weed control relies heavily on tillage.

Research results from USDA-Agricultural Research Service[9] stations in the central High Plains are a main source of literature in this review. To develop wider understanding of the components of integrated farming systems research, including statistical approaches used to interpret results, we also reviewed research activities and results from the Wisconsin Integrated Cropping Systems Trial (WICST), the Long-Term Research on Agricultural Systems (LTRAS) and Sustainable Agriculture Farming System (SAFS) trials in California, the Center for Environmental Farming Systems (CEFS) in North Carolina, and other relevant studies (Table 1)[10–42].

Review of System Components

Irrigated cash crop production

Conversion of dry croplands to irrigated production in the central High Plains in the past 150 years[43] diversified opportunities to incorporate a variety of crop rotations, many of which include annual and perennial forages. There is little research, however, that reports on the relative efficiency of conventional, reduced-input and certified organic production approaches in irrigated systems of the central High Plains. Systems research in irrigated or rain-fed rotations has been documented from the WICST in Wisconsin, the LTRAS and SAFS trials in California, and other projects in California, Colorado, Iowa and North Carolina. Within these systems research frameworks, experiments on long-term agronomic and economic sustainability revealed that alternative production strategies such as minimum-till, no-till and organic farming are more efficient than, or at least as efficient as, conventional practices, especially, in terms of crop production and soil-quality improvement (e.g., [16,20,22,23,30,32,33]).

Cropping system experiments indicate that crop rotations and reduced tillage increase soil organic carbon (SOC) stocks and augment SOC sequestration[22,32,34,35]. In most cases, researchers found higher SOC contents in surface layers of soils under no-tillage systems than soils under standard tillage practices. Higher SOC contents were attributed to changes in the size of microbial communities[32] along with C inputs from crop residues[27]. Reganold et al.[34] reported more soil N, greater microbial biomass, activity and diversity in organically managed soils than conventionally managed soils. In organic and reduced-input systems, alternative soil-fertility inputs such as legumes and organic manures supplied N sufficient to produce yields equivalent to those from conventional systems. However, the timing of N release from organic sources in alternative production systems was different than timing of N release from chemical sources[30]. Alternative management approaches coupled with the application of slow-release N fertilizer are efficient in mitigating greenhouse gas (GHG) emissions from irrigated rotations. For instance, Halvorson et al.[28] found that soils under conventionally tilled corn in eastern Colorado fertilized with standard urea or a polymer-coated enhanced efficiency urea product emitted equivalent amounts of nitrous oxide (N_2O), while soils under no-till corn emitted 49% less N_2O when fertilized with polymer-coated urea than standard urea.

Systems research results show that no-tillage and organic approaches not only improve soil quality and environmental parameters, but also have comparable or higher yields than the conventional production approach. Comparable or higher yields of forage and grains have been observed in organic plots compared to the conventional plots[21,30,33], with the only constraints on organic[33,44] and reduced-input[20] approaches of crop production being increased weeds, insect pests and diseases. High costs of equipment and facilities for higher productivity are characteristic of irrigated systems, and are even more pronounced in organic and reduced-input approaches because weed and insect pest management are more difficult than in conventional production. Soils beneath organic and reduced-input management practices typically have higher microbial biomass and activity[16,34], leading to the possibility of developing opportunistic beneficial or pathogenic associations and, therefore, more frequent disease outbreak if the ecological balance in the soil system changes. Yield reduction and economic loss due to weeds, insect pests and diseases, however, were negligible in organic corn–soybean–oat–alfalfa rotations

Table 1. Production systems, major focus and evaluation (statistical analysis) approaches for existing long-term and short-term integrated farming system studies in North America.

System	Research focus	Analysis[1]	Years	Location	Reference
No-till crop rotations	Long-term productivity and water use efficiency	Model + Univar	12	CO	10
Range land and livestock system	Forage and cow–calf production	Model + Univar	5	CO	11
Mixed grass prairie	Forage production	Model + Univar	18	WY	12
Cropping sequences	Crop yield and SOM	Univar	>100	IL	13,14
Tillage systems, crop rotations and other fertility inputs	Short-term change in SOM and other soil-quality parameters	Univar	2	NC	15
Soil and crop management	Soil microbial community	Univar	7	NC	16
Crop rotations	Crop yield and protein content	Univar	11	Canada	17
Soil-fertility amendments	Crop yield, soil microbes and plant diseases	Univar	10	NC	18
Crop rotations	Crop yield and SOM	Univar	24	Canada	19
Cropping systems and management approaches	Agronomic, economic and environmental effects of pest management	Univar	8	CA	20
Crop rotations transitioning to organic	Soil fertility, crop yield and pest population	Univar	4	IA	21
Cropping systems and management approaches	Annual dynamics of SOC	Univar	12	CA	22
Cropping systems	Nematode dynamics	Univar	5	CA	23
Crop–livestock system	Crop and livestock production, quality, and economics	Univar	4	GA	24
Crop–livestock integrated system—tillage and rotations	Soil physical properties	Univar	4	GA	25
Crop–livestock integrated system—tillage and rotations	SOM dynamics (different fractions)	Univar	3	GA	26
Tillage, cropping system and N fertilization	SOC sequestration	Univar	12	ND	27
Tillage cropping system and N fertilization	N_2O emission, soil mineral N	Univar	2	CO	28
Livestock grazing	Environmental effects, i.e. nitrate leaching and emission to the atmosphere	Univar	3	CA	29
Alternative management systems	N use efficiency and environmental impacts	Univar	9	CA	30
Crop rotations and water supply	Yield and economics of production	Univar	2	CO, NE	31
Cropping systems	Change in SOM	Univar	21	NE	8
Tillage and cropping system	Soil properties, microbial dynamics, plant biomass and crop productivity	Univar + MultVar	2	CA	32
Cropping systems	Crop production and weeds	Univar	⩾8	WI	33
Crop production systems	Soil quality and crop quality	Univar	⩾5	CA	34,35
Alfalfa and other crops	Soil quality and environmental contamination	Univar	3	ND	36
Tillage and cropping sequences	N cycling	Univar	21	MT	37
Tillage and crop rotations	C sequestration and greenhouse gas emission	Univar	1, 5	MT, ND	38
Livestock stocking densities	Livestock grazing pressure index and animal performance	Model	–	ND, KS, CO, WY, SD, OK	39
Cropping systems/intensities	SOC and total N	Univar	12	CO	40
Crop–livestock system	Yield and quality (protein and phosphorus)	Univar	4	ND	41
Crop rotations and fertility inputs	Crop yield and grain protein content, SOM, plant diseases, economics	Univar	18	Canada	42

[1] Univar, univariate statistical approach; MultVar, multivariate statistical approach.

as compared to conventional rotations[21]. Similarly, higher nematode and arthropod populations, as well as more disease-causing organisms, observed in alternative production approaches did not negatively affect yield or farm income[20]. These positive effects might be because highly diverse microbial communities interact through complex food webs to support better growth and productivity of crops compared to the limited microbial diversity in conventional systems. Increased microbial diversity in soils may benefit crops as a result of (i) teamwork between plant-growth-promoting rhizobacteria and *Rhizobium* for improving N_2 fixation; (ii) microbial antagonism for control of plant pathogenic micro-organisms; and (iii) interactions between rhizosphere microbes and arbuscular mycorrhizal fungi to establish a functional mycorrhizosphere[45], all of which ultimately enhance sustainability of agroecosystems.

Transitioning to organic and reduced-input management, however, might affect productivity, and thereby short-term profitability. As the soil environment and microbial processes need to adjust to altered farm inputs and management practices, some researchers have determined that it takes at least 4 years to obtain comparable yields from the alternative management practices[21,46]. Similarly, sugar beet is the basis of profitability on many farms in central High Plains region because its high value offsets high cost of irrigation, but there is currently no market that offers premium prices for certified organic sugar beets. These barriers can thwart transition to alternative management, but they also underscore the need for research into complex, system-scale interactions[13,14].

Integrating livestock in crop production

In central High Plains integrated crop–livestock production systems, maintenance of stock–cow herds for annual production of beef feeder calves for sale to feed lots is the most prevalent practice[47]. Irrigated land produces forage for stock–cow herds that produce calves for sale in the fall and utilize rangelands for summer grazing. Backgrounding, or feeding grain plus corn silage and/or alfalfa hay for several weeks after weaning, is a common practice on farms with irrigated grain/forage crop rotations. Cattle typically utilize cover crops in spring before turnout on rangelands, and forage and grain stubble, supplemented with hay or silage, in the fall and winter.

Integrating livestock into cropping systems benefits agroecosystems through positive effects on soil quality, crop production and environmental services. Integrated crop–livestock system studies across the USA[6,15,18,24,36,48–51] show that integrated production systems benefit producers by (i) more efficiently utilizing farm resources, (ii) exploiting natural pest control processes and/or decreasing the need for synthetic pesticides, (iii) reducing the amount of chemical fertilizer per unit land and minimizing the consequent environmental risks, and (iv) improving soil quality and productivity. For example, comparing tillage systems that included grazing stubbles, Franzluebbers and Stuedemann[26] reported that grazing on no-till croplands increased soil microbial biomass up to 8%, positively affecting nutrient cycling processes and increasing protection of SOC within soil aggregates. In addition, livestock-based systems increase soil biodiversity, reducing occurrence of soil-borne diseases[18,52]. Larger populations of higher trophic level organisms, including macro-fauna such as tunneling dung beetle [*Onthophagus gazella* (Fabricius) and *Onthophagus taurus* (Schreber)], mix crop residues with soil[53]. Increased diversity of soil organisms may also contribute to higher productivity by enhancing mineralization of nutrients from organic resources and increasing the resiliency of agroecological functions and processes (e.g., against drought)[54].

Studies reviewed for this paper consistently show significant benefits of integrating livestock in cropping systems (Table 2), especially in terms of SOC and total soil nitrogen (TSN) enrichment in surface soils. Overall, soils under crop–livestock integrated management had $17.9 \pm 9.4\%$ higher SOC and $12.0 \pm 4.9\%$ higher TSN than soils under conventional, specialized cash-crop production[21,22,40,55,56]; although rates of accumulation varied with geographic location, temperature and precipitation. Similarly, there was $16.6 \pm 0.4\%$ increase in grain yield[21,41] and up to 70.2% increase in stover yield[40] when livestock were part of the system compared to crop-only production systems. The crop–livestock integrated treatments include a range of practices, from in-field livestock grazing to forage production without direct livestock impact. In grazed versus non-grazed treatments, grazing intensity has a strong influence on soil quality[56]. Moreover, integrated treatments responded more effectively to SOC and TSN accrual in the central High Plains than in the southern High Plains and southern Piedmont USA, likely due to lower mean annual soil temperature and slower SOM turnover rates. Soil organic C and TSN decreased by 6% and 1%, respectively, following 2 years of livestock grazing on Cecil sandy loam and sandy clay loam soils with moderately acidic (pH 6) soil conditions of Georgia[26]. This decrease in SOC, although statistically insignificant, might be due to the relatively short time period of the study, a relatively warm (16.5°C mean annual temperature) and moist (1250 mm mean annual precipitation) environment, or high livestock pressure (90% of available forage was consumed in grazed fields). Grain yield decreased by 11% and total aboveground biomass production by 38.4% after these grazing intensity treatments[26,55]. Studies including grazing treatments where SOC, TSN and crop yield responded positively to the livestock-integrated treatment occurred in cooler, drier climates than that of Georgia, and alternative production approaches were evaluated for more than 2 years[55,56]. Moreover, biomass production from pasture varied with grazing intensity: moderate grazing pressure decreased forage yield by only 8.0% compared to

Table 2. Effect of livestock-integrated treatments on SOC, nitrogen and crop performance in different ecosystems across the USA.

Location	Soil type	Sample depth (cm)	Years	Evaluation parameters	Response			Description of the treatments	Ref.
					Livestock integrated	Crops only	% Change		
					----Mg ha^{-1}----				
Watkinsville, GA	Sandy loam	0–30	3	SOC	29.5	31.7	−6.9	[1]Cover crop grazed versus conventional ungrazed system	24–26
				TSN	4.0	4.0	−1.5		
				Grain yield	6.2	7.0	−11.0		
Greenfield, IA	Silty clay loam	0–15	4	SOC	47.4	45.9	+3.3	Four-year rotation (organic grain + forage) versus conventional corn soybean rotation	21
				TSN	4.4	4.0	+7.4		
				Grain yield	5.7	4.9	+16.3		
Eastern Colorado	Fine loam	0–10	12	SOC	12.8	10.6	+20.8	Continuous grain + forage production versus conventional wheat fallow rotation	40
				TSN	1.2	1.0	+18.6		
				Stover yield	2.9	1.7	+70.2		
Cheyenne, WY	Sandy loam	0–30	11	SOC	58.3	48.0	+21.5	Heavy grazed mixed-grass pasture versus ungrazed control	55
				TSN	4.8	4.3	+12.4		
				Aboveground biomass	0.8	1.3	−38.4		
Mandan, ND	Silty loam	0–7.5	9	SOC	25.6	22.2	+15.3	Straw or stover swath grazed during winter versus ungrazed control	41,56
				TSN	2.3	2.1	+9.5		
			4	Grain yield	1.7	1.4	+6.8		
Sacramento Valley, CA	Loam	0–15	12	SOC[2]	11.3	8.8	+28.4	Four-year five-crop rotation using manure + legume/ grass cover crops versus 2-year conventional rotation	22
				TSN	–	–	–		
				Yield	–	–	–		

[1] SOC measured from 0 to 6 cm at the end of second year.
[2] SOC in g kg^{-1}. Text stated that bulk density was equivalent but did not report bulk density or mass per area[22].

ungrazed rotations, but SOC increased significantly under both moderately and heavily grazed conditions compared to ungrazed rotations. The studies reported here indicate that crop–livestock integrated systems support economic and environmental sustainability of agriculture in the central High Plains, but wide heterogeneity in evaluation parameters, along with the limited number of studies that make such comparisons, limit our ability to draw region-wide conclusions and indicate the need for systems research.

Crop–range–livestock systems have been a research priority not only because of their benefits to the soil but also because of their positive effects on crop productivity, farm economy, and ultimately the sustainability and resiliency of farming systems. Integrating livestock in cropping systems increases grain and forage production[41], and also positively affects yield of subsequent crops (Table 2). For example, grazing forage crops rather than allowing them to accumulate as surface residue was found to increase dry matter production of succeeding crops in rotation[24]. In a review, Franzluebbers[50] demonstrated that grazing cover crops may or may not increase crop productivity, but does increase net farm income through benefits such as increasing livestock production and increasing quality of cover crops. The practice benefits farmers and ranchers by increasing efficiency in grazing and manure management, minimizing disease risk by disrupting pest cycles, reducing pressure on natural resources, reducing irrigation water use, increasing crop diversity and diversifying income compared to systems that do not include livestock[57–59]. Similarly, integrating livestock in a wheat–sorghum–fallow rotation on clay loam soils of northern Texas increased productivity of the entire cropping system[60]. In California, removing livestock increased nitrate-N load in downstream spring-fed wetlands[29].

Despite the need for solid data on crop–livestock integrated systems and their common occurrence on farms

of the central High Plains, inclusion of livestock in long-term agricultural system studies is rare. This might be due to logistical challenges associated with integrated research, which requires large areas, many animals, considerable labor and substantial financial resources. Systems research also demands strong interdisciplinary linkage and long-term funding commitments[41,61,62]. Such integrated crop–livestock research is needed, however, in order to improve understanding of environmental impacts and economic trade-offs.

A crucial factor in crop–livestock systems is proper use of nutrients contained in livestock manure for sustainability of concentrated animal feeding operations. Research from other regions shows that continued application of manure nutrients in excess of crop requirements leads to nutrient accumulation, leaching and negative environmental impact[63]. As small- and large-scale animal feeding facilities become more common in the central High Plains, research on manure utilization in dryland and irrigated cropping systems is needed.

A second potentially negative impact of integrated systems that include grazing of forage or crop regrowth, is potential for trampling that can lead to hoof-induced soil compaction[39,64]. Several researchers working in the eastern USA have found that hoof-induced soil compaction in integrated crop–livestock systems negatively affected larger macroaggregates, penetration resistance and water infiltration ability of soils, but did not affect productivity of subsequent crops[24–26,51]. Consistent crop production with or without animal grazing might be because addition of SOM and augmentation of soil microbial activity facilitates mineralization of nutrients and offsets negative effects of compaction. Similar research is needed in irrigated crop–livestock systems of the central High Plains.

An important environmental question about crop–range–livestock systems that needs additional regional research is their effect on C sequestration and their potential to contribute to or mitigate GHG emissions and related effects on climate forcing. Sainju et al.[38,65,66] and Halvorson et al.[28] evaluated GHG emission from dryland and irrigated rotations, respectively, and demonstrated that reducing tillage intensity and increasing N use efficiency help to minimize GHG emissions. Similar research is needed in crop–range–livestock systems, and also in organic and reduced-input management systems, which are becoming more prevalent throughout the USA[67]. Evaluation of CO_2 emissions during manufacture and transport of fertilizer and inputs is important for systems-level evaluation of environmental efficiency of cropping[42] and crop–livestock systems. Liebig et al.[68] emphasized the fundamental role of livestock management in SOC balance in fragile ecosystems of northwestern USA and Canada. However, they found no data on C balance in systems where livestock graze crop residues. They also cautioned that increased N_2O emissions from grazed or manured lands, along with increased CH_4 emission from ruminant livestock, could offset lower net CO_2 emissions from grasslands, possibly resulting in a negative balance on GHG mitigation from integrated systems. Additional systems research should address these questions to help determine practices that optimize synergies in integrated crop–livestock systems. However, maximizing the outputs of such expensive and complex multi-factor systems research requires the ability to evaluate short- and long-term economic and environmental trade-offs with respect to both on-farm sustainability and off-farm ecosystem services.

An Appropriate Evaluation Approach for Agricultural Systems Research

In contrast to reductionistic approaches, research on integrated crop–livestock systems requires evaluation methods that consider all components and interactions, and that draw system-level conclusions with respect to production, environment and economics. Although some agricultural systems studies have compared cropping systems and management approaches in diverse environments[3,6,13,17,19–21,37,40], they typically focus on specific components through reductionistic analyses using univariate evaluation models. But this type of reductionist evaluation system might be insufficient to fully tap information about farming system level problems, complexities, resiliencies and opportunities from systems research projects. A multivariate evaluation system would integrate crop and livestock performance with economic and environmental efficiency variables on whole-farm or agroecosystem scales[69]. Multivariate approaches for whole-system evaluation could help to define production, logistical, and environmental limitations and benefits of integrated crop–livestock production systems[50].

Several researchers[10–12,70] have developed decision support tools for evaluating sustainability of such cropping systems in the central High Plains. Similarly, the Great Plains Framework for Agricultural Resource Management (GPFARM) forage growth model[12] was developed to predict stocking rates for both strategic (long-term) and tactical (within-season) grazing management in northern mixed-grass prairies. Integrating cropping system models with forage and livestock production models might provide opportunities to develop decision support tools in integrated settings. Several other studies[39,71,72] have reviewed the performance of integrated crop–livestock systems in the central High Plains and revealed interactions among farming system and environmental components. For example, Hendrickson et al.[71] evaluated the interaction among cropping systems and environment in North Dakota, South Dakota, Nebraska and Kansas, and was able to draw broad conclusions about how environment shapes, and is shaped by, integrated farming practices. In such studies, models are mainly used as decision support tools for adjusting

various parts of multicomponent systems. As such, they are important for optimizing management activities, but do not serve to evaluate system performance and interactions among many different variables.

Performance of complex integrated agricultural systems can be evaluated through multivariate, multifunctional evaluation models that might maximize return on investment in expensive and complicated long-term systems research. Tanaka et al.[72] presented a framework for dynamic cropping systems that utilizes a variety of soil and plant management practices with a diversity of crop species to reduce risks of disease, weed and insect pest outbreaks in the central High Plains. Reganold et al.[34,35] evaluated productivity, economics and environmental sustainability of apple production in Washington State and strawberry production in California, and demonstrated the benefits of using interdisciplinary methodologies that comprehensively and quantitatively compare indices of crop performance and soil quality from multiple farms, multiple varieties and soils, and multiple sampling times. Robertson et al.[3] called for creation of a national network of long-term agricultural systems research sites with standardized monitoring and multivariate evaluation procedures across many agroecosystems. This system would be modeled on the successful National Science Foundation Long-Term Ecological Research (LTER)[73] network and could pool resources to optimize information that could be realized from expensive research efforts. Creation of such a network, along with development of suitable statistical evaluation models to evaluate existing long-term integrated experiments, could raise our understanding of complex agricultural systems just as the LTER network has elucidated many complex, multifactor ecological questions.

Implementing Agricultural Systems Research in the Central High Plains

Soil properties typically respond slowly to management changes in the central High Plains because cool, dry conditions limit rates of chemical and biological processes[74,75], plant productivity and associated SOM processes[50,75]. West and Post[76] indicated that improved management practices such as reduced fallow, reduced tillage intensity, increased crop diversity and application of integrated agricultural practices support farmers by improving economic and environmental sustainability. While some of these practices show promise in the central High Plains, relatively low productivity, harsh and highly variable climate, and long distances to markets for both inputs and products create unique challenges in this region. Integrated crop–livestock systems include promising options to improve economic and environmental sustainability of central High Plains agroecosystems, but there is much yet to learn to fully understand how management of system components might improve efficiency and environmental services in integrated crop–range–livestock systems[77]. Long-term integrated agricultural research could support development of resilient systems in the context of pressing need for higher profitability and stability of farm income, growing concerns about environmental change and increasing regulations[49]. Most of the holdings in central High Plains are small to medium sized, with annual sales of less than US $500,000[49,78]. Diverse, integrated operations can reduce risk in the face of increasingly unpredictable climate, prices and costs[68,79].

Integrating livestock into cropping systems may represent an important option for crop production specialists in the face of changing climate in the central High Plains, as rising temperatures and limiting moisture shift the best choice of crops toward perennial forages[80]. Soils of the central High Plains region are regularly subjected to wind erosion, and there is relatively little crop residue recycling. Perennial forage crops, with deep roots and more efficient water and nutrient use, are better adapted than annual crops to projected climate changes in this region. Systems that include livestock include options such as cutting grain early for use as forage if water runs short, weeds are too prevalent, frosts force late planting, or other problems affect grain development and maturity of crops. Similarly, more efficient feed and manure use reduces manure oversupply through improved internal cycling of nutrients, ultimately reducing GHG emissions[81]. Eagle et al.[82] discussed the potential of US agricultural lands to mitigate GHG emissions through adoption of alternative management options. Evaluation of GHG emissions are, however, lacking from central High Plains agricultural systems[83].

Conclusions

In this review, we evaluated agricultural systems research in the central High Plains and revealed distinct impacts of alternative management practices that include reduced tillage, certified organic management and integrating livestock into cropping systems on SOC, nutrients and crop production. Building of SOM, however, is a slow process in this region because of the cool, dry climate and limited biomass recycling[74]. In this scenario, implementing integrated crop–livestock systems may represent an important option toward enhancing agricultural resiliency in changing climatic and economic conditions. Most interdisciplinary studies, however, focus on specific aspects rather than whole systems, and there is a need for integrating productivity, profitability and environmental sustainability variables in a single evaluation frame in order to effectively generate information toward enhancing the adaptability of farming systems. Similarly, there is a need for a suitable multivariate evaluation system to reveal primary variables that drive the strengths and weaknesses of alternative production systems, especially in the highly variable environment of the central High Plains.

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Corrigendum
Diversification and ecosystem services for conservation agriculture: Outcomes from pastures and integrated crop-livestock systems – Corrigendum

Matt A. Sanderson, David Archer, John Hendrickson, Scott Kronberg, Mark Liebig, Kris Nichols, Marty Schmer, Don Tanaka and Jonathan Aguilar

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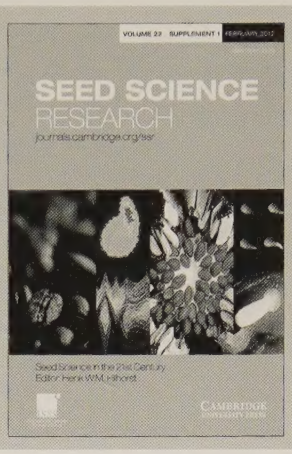

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